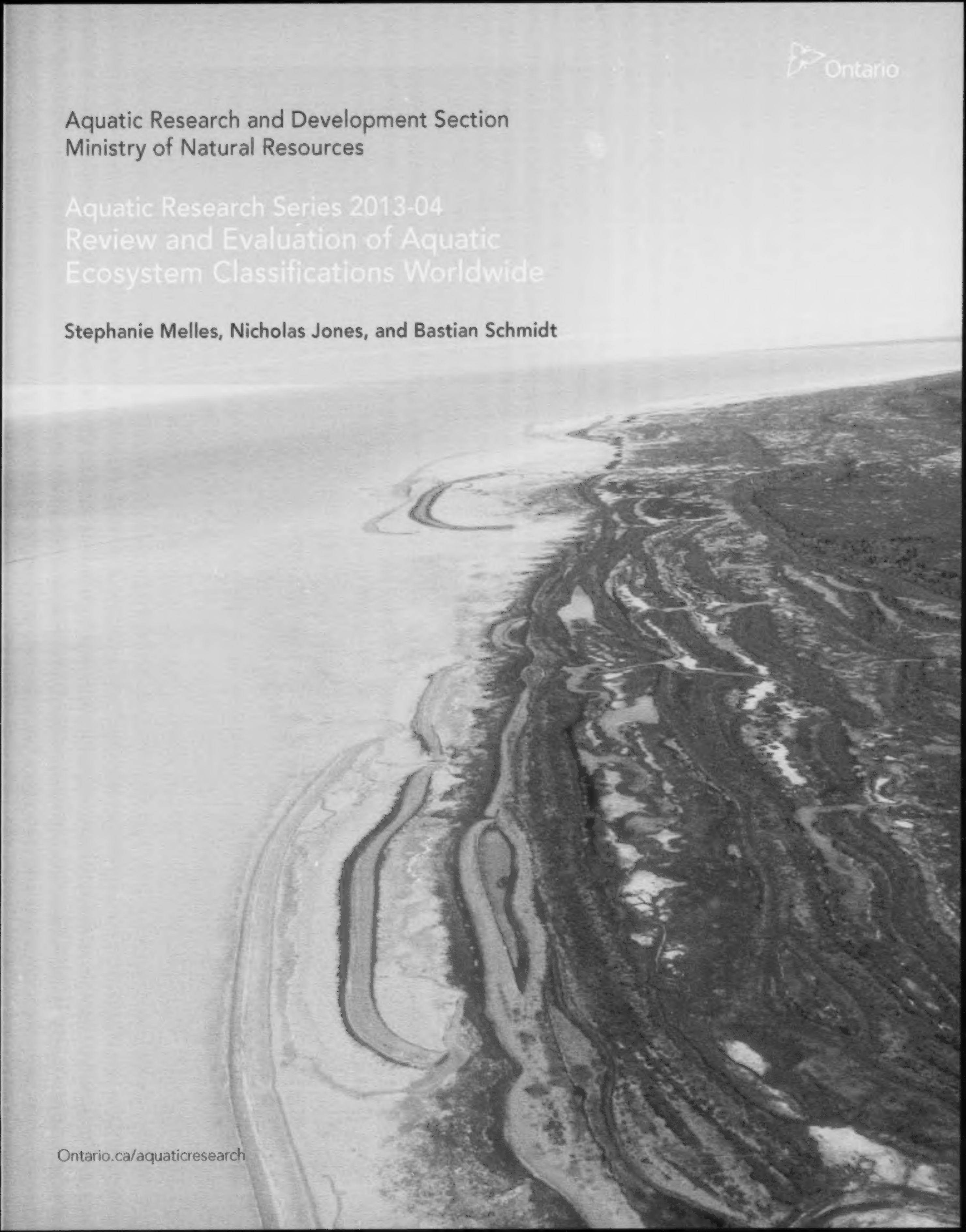

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Review and Evaluation of Aquatic Ecosystem Classifications Worldwide

Stephanie Melles, Nicholas Jones, and Bastian Schmidt

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Executive Summary

Aquatic ecosystem classifications (AECs) help governments manage and conserve aquatic resources at regional, provincial, and national scales. A variety of stream and lake classification methods are available, and such methods attempt to provide a systematic approach to modelling and understanding complex aquatic systems at various spatial and temporal scales. However, a generally accepted approach to aquatic classification has not been produced. This report provides a review of the literature on AECs with a focus on more recent classifications since 1980. The majority of aquatic classifications consider different types of aquatic systems separately (i.e. lakes, wetlands, groundwater aquifers, and streams are classified separately). Our focus was on stream and flowing water classifications, including those that attempted to integrate all elements of aquatic networks, such as lakes and wetlands where possible. *Section 1* introduces the different types of classifications including abiotic (geoclimatic, geomorphic, hydrologic, and chemical) and biotic classifications (e.g. based on fish and benthos). This section also summarizes some of the progress that has been made by the OMNR on aquatic classification in Ontario since 1995. Published literature reviews of aquatic ecosystem classifications are summarized and highlighted. *Section 2* covers differences between terrestrial and aquatic classifications and cites a considerable body of research that demonstrates land-based classifications have limited application for aquatic bioassessments. *Section 3* summarizes results from a quantitative assessment of AECs in use around the world and this information forms the basis of a journal article. The assessment looks at variables used in classifications, their approach to classification, and the scales of analysis, finding that the majority of classifications have multiple levels (stream reach, segment, watershed, whole system, and ecoregion) and use variables that relate to geology, gradient, and climate. *Section 3* also provides a more in depth evaluation of classifications used in Ontario and the United States. *Section 4* provides a synthesis of known limitations to aquatic classification and directs the reader to the draft proposal for Aquatic Ecosystem Classification in Ontario.

Résumé

Les classifications des écosystèmes aquatiques (CEA) aident les autorités à gérer et préserver les ressources aquatiques à l'échelle régionale, provinciale et nationale. Il existe diverses méthodes de classification des cours d'eau et des lacs et ces méthodes visent à fournir une approche systématique à la modélisation et l'entendement des systèmes aquatiques complexes à diverses échelles spatiales et temporelles. Cependant, une démarche généralement reconnue pouvant servir à la classification aquatique n'a pas encore été produite. Dans ce rapport, on trouvera une analyse de la littérature sur les CEA avec une concentration sur les classifications récentes depuis 1980. La majorité des classifications aquatiques rendent compte séparément de différentes catégories des systèmes aquatiques (c.-à-d., les lacs, les terres humides, les nappes souterraines, et les ruisseaux sont classés séparément). Notre attention portait principalement sur les classifications visant les cours d'eau et les eaux mouvantes, et notamment celles qui ont tenté d'intégrer tous les éléments des réseaux aquatiques, tels que les lacs et les terres humides autant que possible. Dans la section 1 sont présentés les différents types de classifications, notamment les classifications abiotiques (géoclimatiques, géomorphologiques, hydrologiques et chimiques) et les classifications biotiques (p. ex., basées sur les poissons et le benthos). Cette section propose aussi un survol des progrès qui ont été réalisés par le MRNO relativement à la classification aquatique en Ontario depuis 1995. Les analyses de la littérature publiée sur les classifications des systèmes aquatiques sont résumées et mises en relief. La section 2 porte sur les différences entre les classifications terrestres et aquatiques. On y cite une somme importante de recherches ayant démontré que les classifications en fonction du sol ont une application limitée pour les bioévaluations aquatiques. Dans la section 3, on trouve la synthèse des résultats d'une évaluation quantitative des CEA qui sont utilisées dans le monde entier et cette information forme la base d'un article publié dans un périodique. L'évaluation examine les variables utilisées dans les classifications, leur approche à l'égard de la classification, et les niveaux d'analyse, pour conclure que la majorité des classifications comportent de multiples niveaux (tronçon de cours d'eau, segment, bassin hydrographique, système entier et écorégion) et utilisent des variables qui se rapportent à la géologie, à la pente et au climat. La section 3 présente aussi une évaluation plus approfondie des classifications utilisée en Ontario et aux États-Unis. La section 4 présente le bilan des limitations connues associées aux classifications aquatiques et dirige le lecteur vers l'avant-projet de Classification des écosystèmes aquatiques en Ontario.

1 Introduction

At their most basic level, classifications are designed to order objects into meaningful groups on the basis of similarities, differences or relationships that people deem to be important (Sokal 1974, Portt et al. 1989, Naiman et al. 1992, Cleland et al. 1997). There have been many attempts to classify freshwater ecosystems; but equally many have failed to be fully adopted or successfully implemented (Pennak 1971, Portt et al. 1989, Hudson et al. 1992, Soranno et al. 2010). Although it is difficult to pinpoint exactly why a classification system is not supported, freshwater classification systems face at least two key challenges (Soranno et al. 2010). First, ecosystem classes are ideally based on known empirical relationships between the drivers of change and characteristics of freshwater systems (Young et al. 2002, Soranno et al. 2010, Melles et al. 2012), but these relationships are not always known. Classification systems are effectively a reflection of the current state of knowledge on river function (Frissell et al. 1986). Second, the management objectives for a classification system need to be tied to the design or associated end points (e.g. maps and categories, Soranno et al. 2010). Aquatic classifications can help satisfy a variety of management goals and objectives (Box 1.1, Soranno et al. 2010). But, no classification can serve all of these functions. Care must be taken to ensure that management and decision makers are involved in the design of the classification system intended for their use. The end-user community must be ready to understand, adapt, and support the continued use of any new aquatic ecosystem classification.

Essentially, all classificatory efforts aim to satisfy the following four general principles. First, classifications organize information and describe what is known. The hope is that the description captures the true nature of the objects or relationships described. When classes reflect the actual processes that have led to the groupings themselves, then by studying the classification, we can learn about the laws governing the behaviour of the objects classified (Sokal 1974, Portt et al. 1989). This allows us to make inductive generalizations about objects in each class (Portt et al. 1989). Second, classifications achieve what Sokal (1974) called, an ‘economy of memory’. Just as language allows us to attach convenient labels to things to aid communication, classifications provide a consistent spatial context for monitoring, inventory, reporting, and reference. Third, classifications are easy to understand and manipulate such that new objects can be easily classified. Fourth, classifications represent hypotheses about how we think ecosystems behave. They stimulate interest and raise questions about how the perceived order was generated (Sokal 1974).

Box 1.1. Management goals & objectives for Aquatic Ecosystem Classification

1) **Platform for landscape level planning** and resource management
    a. Strategic planning for sustainable development
    b. Conserve biodiversity (i.e. habitat and species)
        i. Set priorities for protection
        ii. Set priorities for representation of diversity
    c. Set rehabilitation and restoration targets

2) **Tool for science development**
    a. Select reference ecosystems
    b. Establish baseline
    c. Make predictions
    d. Assist in the development of indicators and standards

3) **Framework to assess** State of Aquatic ecosystems and resources
    a. Assist with directing cost effective sampling efforts
    b. Provide a consistent spatial context for;
        i. Inventory
        ii. Monitoring
        iii. State of the Environment Reporting
        iv. Pollution assessments

1.1 Background – AEC development in the province of Ontario

A considerable amount of work has already been done in the province of Ontario towards the development of an aquatic ecosystem classification (AEC). Workshops on AEC with a focus on flowing waters were hosted by the OMNR in Maple (1995), Toronto (1998), and Peterborough (2008). One of the key ideas emerging from the workshop in 1995 was an identified need for a hierarchical classification system for lakes, rivers, wetlands and ground water. Participants recommended that an AEC be supported by the OMNR and developed in parallel to the terrestrial classification system. A set of next steps were identified including the establishment of a classification working group with the following tasks: prepare an implementation plan, develop a conceptual model for classification, and verify appropriate management applications. Background documents included: an overview and report on the ecological land classification program (Perera et al. 1994, 1996); a report on the hierarchical process of aquatic classification in the U.S. at the time (Maxwell et al. 1995); and, a review of stream habitat classification by Portt et al. (1989).

In 1998, the first step towards finalizing a framework for *"Classification and Protection of Ontario's Aquatic Natural Heritage"* was created at a workshop in Toronto by the then classification working group. The proceedings from this workshop (McGovern and Moore

1998) document the results of a three-day experts' session on developing an Aquatic Natural Heritage - Classification System (ANH-CS) for OMNR. A companion document (Mandrak 1998) presents the results of an application to test an existing *terrestrial* eco-regionalization to determine its' suitability for aquatic ecosystems. Workshop proceedings identified several common elements among existing aquatic classifications in the literature pertaining to applicable data layers and the principle driving forces of aquatic diversity (e.g. the geoclimatic setting, hydrology, species distributions, physiography (i.e. physical geography such as land formations, clay plains, kame moraines), water chemistry, soils, gradients in elevation, channel form, stream size, and surrounding land use). Information needs at the time were identified.

Mandrak (1999) worked further on a framework for an AEC in Ontario and identified the upper levels in a spatially nested, hierarchical classification that included aquatic ecozones, aquatic ecoprovinces, and aquatic ecoregions. The approach was focused on establishing a correspondence between abiotic aquatic eco-regional groupings (i.e. based on bedrock geology, climate, glacial history and surficial geology) and fish faunal regions. Though Mandrak proposed a framework to classify mid- and lower levels of a hierarchical AEC, he identified only the uppermost (broad-scale) levels. Lower levels, according to Mandrak, should be based on variables related to local aquatic community patterns (e.g. stream flow, substrate, depth, cover) and groupings of hydrologically defined entities (e.g. watersheds, lakes, valley segments, stream reaches). Work on Mandrak's framework and classification in Ontario was not directly pursued.

The Blueprint project was a joint initiative between the Nature Conservancy of Canada (TNCC), OMNR, and the Natural Heritage Information Centre (NHIC), which began in 2001 with the goal of producing aquatic and terrestrial conservation targets in the Great Lakes region of concern (Phair et al. 2005). A key component of the Great Lakes Conservation Blueprint project was an aquatic ecosystem classification for the region (Wichert et al. 2004, 2005). This AEC was largely based on an approach proposed and used by the U.S. Nature Conservancy (e.g. Higgins et al. 1998, Groves et al. 2000, Higgins et al. 2005). The Nature Conservancy of Canada is currently pursuing this approach to aquatic classification in British Columbia. See for example: Link.

In November 2008, a workshop on developing an Ontario-wide AEC was held (Jones 2008). The goal of the workshop was to identify future directions for AEC in the Province. The workshop was attended by various representatives from OMNR branches, the Department of Fisheries and Oceans, academics, and consultants. In 2009 and early 2010, a workshop summary report and literature review (Wichert 2010) were produced by AECOM to support the development of an aquatic classification. The proposed classification would focus on the abiotic template that filters species, not the species themselves. Abiotic variables are easier to measure remotely and this should facilitate mapping of regions with inadequate field data (e.g. the Far North). Four spatially nested hierarchical levels were identified. Proposed future steps involved: a literature review; clarification of the purpose and objectives of an AEC; identifying data and information needs that support key approaches to classification; developing data organization and presentation tools to allow analyses of data in ways that suit users; creation of a user needs assessment; development of an evaluation method; and identification of a technical team, including managers and those with experience in classification systems. Workshop participants emphasized that management objectives need to be clearly articulated.

As this *Section* indicates, there have been various attempts made towards creating an Provincial AEC within the OMNR. The purpose of the present review is to provide a thorough evaluation of different approaches to classification in Ontario and around the world. The Ontario-based aquatic classifications by Mandrak (1999) and Wichert et al. (2004, 2005) are evaluated in more detail in *Section 3.2*. This review builds on past efforts towards the development of an AEC in Ontario and attempts to shed some light on the way forward towards a successfully implemented AEC.

1.2 Types of classifications

A grouping of streams based on a ***single variable***, such as tributary-confluence breaks, is the simplest form of aquatic classification (e.g. Gravelius 1914, Horton 1945, Mock 1971). Most modern classifications are based on ***multiple variables*** (i.e. flow, current speed, substrate, depth, width, temperature, chemistry, vegetation) and are much more complex than this (e.g. Pennak 1971). Complexity is generally addressed using a ***hierarchical*** approach in which the entire system is subdivided into groups based on various criteria that determine how objects are arranged into successively finer levels of detail. But, single level classifications of all similarly sized aquatic units are also possible (e.g. Seelbach et al. 2006, Hakanson et al. 2007).

Different types of hierarchical aquatic classifications have been around a long time – at least since the 1920's (Carpenter 1928, Portt et al. 1989, Uhlig & Baker 1994), and the conceptual framework of creating levels of organization in a hierarchical fashion dates back to antiquity (Wu & David 2002). There are three main approaches to hierarchical classification that can be found in the literature on aquatic ecosystem classification; 1) bottom-up, 2) top-down, and 3) dual top-down, bottom-up approaches. Bottom-up approaches are traditionally ***place-independent*** wherein geographic location is irrelevant to the classification of a site (Bailey et al. 1978, Hudson et al. 1992). For example, sections of stream that flow over bedrock with an average slope of 5% somewhere on the Canadian Shield might be grouped with sections of stream found in north-eastern Minnesota based on the same characteristics (e.g. Rosgen 1994). A stream of a given type in place-independent classifications can be found anywhere in the world. These classifications follow the classical approach to grouping objects into successively more detailed hierarchical strata based on various criteria, and the resulting groups are spatially contiguous.

Spatially-nested classifications are ***place-dependent*** or ***regionalized*** and they can also be created from the bottom-up by making groupings of aquatic ecosystem units under spatial contiguity constraints (e.g. Brenden et al. 2008a, Snelder et al. 2010). But regionalization, more generally, is the process of subdividing a unit of land according to observed patterns of variation from the top-down (Bailey et al. 1976, Portt et al. 1987, Bryce & Clarke 1996). In practice, the upper levels of regionalized hierarchical classifications are often essentially sketched on the basis of expert opinion and local knowledge of ecosystem characteristics

(Omernik 1987, Kleynhans et al. 2005, Higgins et al. 2005, Seelbach et al. 1997, 2006). Lower level classes at finer spatial scales within regionalized classifications are often created through an iterative process involving top-down and bottom-up approaches, and the result is a mixture of regionalized and place-independent classes at a variety of scales (e.g. Wichert et al. 2004). Classes may be formed on the basis of selected nominal attributes with expert- and scientifically- driven cut points (e.g. large, medium, or small lakes) in top-down or bottom-up classifications (e.g. Higgins et al. 2005).

Explanatory variables used in aquatic classification can be ***abiotic, biotic***, or ***both***. ***Abiotic*** classifications incorporate physical and chemical characteristics including numerous parameters that attempt to describe the physico-chemical characteristics of different species' habitat, often at a variety of spatial scales. There are at least four different types of abiotic classifications: ***geoclimatic, geomorphic, hydrologic, and chemical.***

1. ***Geoclimatic*** classification systems use climate, physiography and surficial geology as the fundamental basis for classification. These classifications are based on the premise that streams draining areas with similar climate and geology will have similar biota because climate and geology are the most important drivers of species' habitat and dispersal (Lotspeich and Platts 1982, Portt et al. 1989).

2. ***Geomorphic*** classifications take a structural engineering approach to classification and are widely used at both the reach (Rosgen 1994, Montgomery & Buffington 1997) and valley-segment scales (from confluence to confluence; e.g. Frissell et al. 1986, Maxwell et al. 1995, Brierley & Fryirs 2000). The basic premise of geomorphic classifications is that channel morphology is governed by the laws of physics through fluvial processes and readily observable features of stream channels (i.e. valley and reach slope, substrate characteristics, sediment supply and transport, constraint by valley walls, presence of pools/rifles, Rosgen 1994, Montgomery & Buffington 1997).

3. ***Hydrologic*** classifications use numerous indices of flow to classify streams, including; measures of flow variability, stream discharge, current speed and runoff, mean daily, seasonal, or annual flow, groundwater levels, baseflow, and water source (Kim & Hawkins 1993, Poff and Ward 1989, Zhang et al. 2011). Many authors consider flow to be a "master variable" because it is strongly correlated with many abiotic and biotic characteristics of interest in stream classification (Poff et al. 1997, Gordon et al. 2004). For example, flow is related to both channel geomorphology and water temperature, and flow can limit the distribution and abundance of many riverine species thereby regulating 'ecological integrity' (Poff & Ward 1989, Poff et al. 1997, Jones 2008, Wichert 2010).

4. ***Chemical*** or water quality based classifications are often designed to assess the physical and chemical characteristics in streams according to criteria established by various environmental protection agencies (e.g. Ahl 1994, Dodds et al. 1998, Gordon et al. 2004). The criteria of interest attempt to capture the degree of eutrophication, as reflected by nutrient levels such as nitrogen and phosphorus, and toxicity of stream

pollutants that negatively affect human and ecosystem health (e.g. pH and toxins, Scottish Environment Protection Agency, SEPA 2009). Water quality classes are typically indicative of their suitability to certain fish species (Gordon et al. 2004).

In contrast, ***biotic***-based classifications group observational units predominantly on the basis of the presence of various taxa such as: game ***fishes*** (Shelford 1911,Wicker 1934, Huet 1959, Illies 1961, Anonymous 1967), ***benthic invertebrates*** (Knight & Gaufin 1967, Wright et al. 1984, Biggs et al. 1990), ***aquatic macrophytes*** (e.g. used by Pennak 1971 and others), and ***riparian vegetation*** (e.g. used by Pennak 1971 and others). Biotic classifications are appealing when the objective is to manage the biota (e.g. fisheries resources), or when detailed data on macroinvertebrate aquatic community assemblages are available (Wright et al. 1984, 1989, 2000), but classifications based on sport or game fishes alone will likely be inadequate for most integrative purposes (Portt et al. 1989). Most authors realize that both abiotic and biotic factors contribute to community structure at various times (Poff & Ward 1989 and references therein). Yet, determining the relative importance of abiotic and biotic factors in time and space for individual species in the (aquatic) community is one of the most challenging problems of ecology today.

1.3 Other resources

A number of other reviews of aquatic classifications are available, which the reader may wish to consult. In the secondary literature, a few unpublished reports with particular relevance for Ontario are worth mention. Portt et al. (1989) provide a very good review of the primary and secondary literature on stream habitat classification systems up to 1989. They group stream classifications into three types: abiotic, biotic, and geoclimatic and they conclude that a geoclimatic stream classification system would be most appropriate for southern Ontario. Jones (2008) provides a review as a companion document for the workshop on AEC held in Peterborough by the OMNR (fall 2008). He develops a rationale for AEC in Ontario and briefly covers some of the key literature. Certain variables to consider were identified (e.g. conductivity, turbidity, hydrothermal regime). The Jones report also makes a link between variables that constrain stream biota and variables that constrain sampling methodologies. Wichert (2010) provides a condensed review of the literature on stream classifications to support development of an AEC for Ontario as a follow up to the 2008 workshop held by the OMNR. According to Wichert, many classification systems are consistent with key concepts in ecology and many of these key concepts are linked to spatial scale. The section on hydrologic flow may be particularly useful because it touches on important studies and trends. For example, Wichert highlights the fact that stream hydrology is considered to be a master variable that regulates ecosystem integrity (Poff et al. 1997). Other influential hydraulic variables of note were: flow per unit drainage area; coefficient of variation of mean annual flow; and flow constancy.

In 1989, Ryder & Pesendorfer published a comparative review of large rivers and flowing lakes in the proceedings of an international large river symposium. They compare and contrast differences between large lakes and rivers of the world under the context of the river continuum concept (RCC). Their focus is sequentially on morphology, hydrodynamics, temperature, chemistry, and ecology. Comparison indices of each of these factors are presented in tables for rivers and lakes. In terms of morphology, Ryder & Pesendorfer found that slope (or gradient) is often indicative of the type of fish fauna present, and they suggest that rivers of comparable breadth, depth and slope should have similar biotic characteristics.

Goodwin (1999) provides an overview of the classificatory endeavour with a focus on a review of (geomorphic) fluvial classifications in North America. He recognizes three basic types of classifications after Smith & Medin (1981): 1) classical classifications have members that are defined by common properties such that all members have equal status and no members are more representative than other members of that class; 2) probabilistic classifications do not require all properties to be true of all class members; and 3) exemplary classifications use representative individuals as exemplars that define the class. Goodwin further divides classification schemes according to whether they define *nominal* groups for a specific purpose or whether they objectively find *natural kinds* or unbiased breaks in the (aquatic) environment. He reasons that the latter group alone should provide an aid to scientific understanding because they will be based on processes and causes rather than descriptive convenience. Goodwin goes on to provide 10 recommendations for improving fluvial classification systems suggesting in particular that they be based on processes or controlling factors (e.g. discharge, sediment transport, deposition and erosion); that they include temporal change and *"processes through time"*; that they be probabilistic, based on sound theory, and treated as hypotheses rather than paradigms.

Several book chapters are also worth reading. The proceedings of a 1988 symposium on classification and inventory for Great Lakes aquatic habitats were published in 1992 in a book edited by Busch & Sly (1992). Three chapters merit discussion: a chapter on classification schemes appropriate to streams and rivers by Hudson et al. (1992); a chapter on Lake (habitat) classification by Leach and Herron (1992); and a chapter on wetland classification systems by McKee et al. (1992).

1) Hudson et al. (1992) provide a review of 70 stream habitat classification papers written between 1911 and 1987. They found that eighty-five percent of the most important descriptors of stream habitat in studies examined were made up of geomorphological attributes (gradient, velocity, substrate, stream order, flow, volume). Half of the examined studies used place-independent classification methods and the remaining 50% used place-dependent methods. Hudson et al. (1992) grouped stream and river classification systems into four types: longitudinal zonation schemes (including 39 descriptive classifications from the period 1911 to 1987), large-river schemes (6 studies between 1942 and 1989), large-scale physiographic schemes (9 studies between 1971 and 1987), and predictive schemes (17 studies between 1973 and 1987). An emphasis on predictive schemes was evident in the more recent studies.

2) Leach & Herron (1992) provide a comprehensive review of approximately 140 articles covering lake classification systems. Lakes can be classified based on a variety of characteristics, which can be grouped into five basic types: 1) lake origin, shape, and regional location; 2) physical properties of lakes (e.g. thermal mixing and their optical properties for remote sensing applications), 3) chemical properties (e.g. edaphic inputs and water quality), 4) trophic status (e.g. oligotrophic, mesotrophic, and eutrophic), and 5) fish habitat and the assemblages that they support. By far the most common

classification schemes in use up to the time of their review were lake trophic classifications designed to quantify or describe the factors contributing to accelerated lake eutrophication. Leach and Herron conclude their review by suggesting that a regional classification of lakes may be the best approach to provide a first-order inventory of fisheries habitat. They recommend a hierarchal approach to fish yield estimation with global, regional, and infra-regional levels determined by driving variables such as temperature, area, nutrient inputs, and lake morphometry.

3) McKee et al. (1992) thoroughly review three wetland classification systems with established application in the Great Lakes region including a national system (Cowardin et al. 1979), a regional system (OMNR/CWS 1984), and a system developed specifically for Great Lakes coastal wetlands (Jaworski & Raphael 1979). Wetlands were defined as transitional areas between terrestrial and aquatic systems with a water table close to or at the surface and covered by shallow water either seasonally or permanently. McKee et al. conclude that the wetland and deepwater aquatic system classification proposed by Cowardin et al. (1979) would be most suitable for application in the Great Lakes area with some modifications to incorporate water-level regime and storm-surge modifiers.

In northern Ontario, a technical report compiled by John Riley (2011) describes wetland types in the Hudson Bay lowlands. The lowlands of Ontario support the largest wetlands in North America and these are among the third largest in the world. Eighty-five percent of the Hudson Bay lowlands are mineral wetland or organic peat land. Riley (2011) provides an analysis of ecological variation and succession in wetlands as well as a catalogue of available wetland field data collected in the region between 1972 and 1990. Field data include vegetation samples ($n = 309$) and abiotic information such as surface water pH, temperature 10 cm below surface water level, substrate type, peat depth and elevation.

Additional book chapters by Naiman et al. (1992, 1998), and Gordon et al. (2004) make worthwhile reading. Naiman et al. (1998) focus on biotic stream classifications, evaluating their relevance to conservation assessment and management. But they also summarize historical concepts and hierarchical approaches to watershed classification at multiple and nested spatial scales (Naiman et al. 1992, 1998). These physical classification models are akin to terrestrial approaches (Bailey 1978, Frissell et al. 1986) and they rank geology and climate as the driving variables at broad spatial scales. The importance of biotic community data is emphasized by Naiman et al. (1998) as they provide an indication of both environmental quality and conservation potential. However, in order to collect data on biotic communities, extensive expertise and intensive field work are required. Moreover, application of systems based on biota is limited because biotic communities change so substantially across different regions of the globe.

Gordon et al. (2004) thoroughly review the use of stream classification in management applications and theirs is an excellent source for many of the topics covered in the present work. These authors cover the purpose and principles behind classificatory exercises, and they review the theories and concepts that have structured thinking around lotic environments. Ten types of stream classifications for management purposes were identified including: those based on 1) the energetics, structure, function and dynamics of rivers; 2) geomorphic classifications; 3) hydrologic classifications; 4) water-quality based classifications; 5) physical-chemical-ecological classification models; 6) ecoregional and multi-scale classifications; 7) wetland and 8) estuary classifications; 9) classifications based on the conservation value of rivers; and 10) designated use classifications.

More recently, Thorp et al. (2008) wrote a book entitled, *Riverine Ecosystem Synthesis*, which outlines a general theory behind physical models of repeating watercourse 'patches' or functional process zones (FPZs). Although still in development, they outline a framework for the study, management and conservation of river 'landscapes' (FPZs) based on the view that rivers contain patches with predictable environmental characteristics that are formed by dynamic and hierarchically structured processes operating at a variety of spatio-temporal scales.

A final important resource related to the present review is the recent paper by Melles et al. (2012). These authors examine theoretical developments in stream ecology and geomorphology over the last century to understand how scientific developments in these fields have influenced stream classification and conservation planning for flowing waters. They separate the periods of conceptual influences into three overlapping phases distinguished by theoretical, analytical or technological advances: I) early Darwinian perspectives; II) the quantitative revolution; and, III) age of the computer, hierarchy and scale. Early ideas that streams progress deterministically through successive stages of evolution seemed to create a veil that barred analysis of the full scope of variability in these systems for over 50 years. The quantitative revolution focused on predictive and mechanistic explanations of stream processes, setting the stage for physically-based stream classifications that assume streams can be restored by engineering their physical characteristics. Concepts from the fields of GIS and landscape ecology have been incorporated into stream ecology and aquatic classification during the 'age of the computer', hierarchy and scale. However, fully integrated aquatic ecosystem classifications that incorporate lake and river networks, wetlands, groundwater reservoirs and upland areas are exceedingly rare. They reason that the way forward for classification of flowing waters is to account for the directionally-nested nature of aquatic networks and to encode flexibility into modern digital freshwater inventories and fluvial classification models (Wang et al. 2011).

1.4 Outline of this review

One would be hard pressed to find a comprehensive review of a subject area today that can be considered truly complete. There is simply too much literature and information on most subject matters available. In this document, we have made a reasonable attempt to review pertinent primary and secondary literature in the subject area.

- In *Section 2*, we describe the terrestrial approach to land classification in Canada and the United States. The suitability of terrestrial classifications for making inference in aquatic systems is examined by reviewing a number of studies that test the performance of regional terrestrial classifications for their ability to partition biotic variability among aquatic sites. The need for a separate AEC is discussed.

- *Section 3* describes work on aquatic ecosystem classifications around the world, including approaches taken in Canada, the U.S., New Zealand, Australia, the European Union, and South Africa.

- In *Section 4* we discuss advantages of aquatic classifications in addition to management applications.

- *Section 5* outlines known limitations, such as they are, regarding; the potential complexity of AECs, dealing with spatial and temporal variability, biotic processes (e.g. species interactions), indistinct boundaries, lateral and vertical connectivity, and the need for testing classificatory systems. The review closes with some brief conclusions; recommendations for the road ahead can be found in a related technical report that outlines a framework for AEC in the Province of Ontario (Melles et al. *draft*).

2 The terrestrial approach

2.1 Terrestrial classifications in Canada and the U.S.

There is a long history of ecoregional land classification in the United States (Bailey, 1985) beginning with Allen (1892), Merriam (1898), and Dice (1943). Dice (1943) based maps of biotic provinces in North America on the biome concept, which refers to large land formations or units with a particular climate and faunistic community (Clements & Shelford 1939). Terrestrial ecological regionalizations fall under the same category as biogeographic regions, ecoregions, environmental domains, ecological land units, land classes, and environmental classifications (Snelder et al. 2010). An ecoregion can be defined as a relatively homogeneous, ecologically distinctive area that has been shaped by a combination of factors such as geology, landform, soil, climate, water and biota (i.e. vegetation, wildlife, humans, etc.). As the biota can change the ecology of a region, variables like vegetation and human land use patterns can be interpreted as both response variables and transformative ecosystem process variables.

In 1965, Austin produced a report and map of '*Land Resource Regions and Major Land Resource Areas of the United States*'. This classification was largely a revision of previous maps showing 'problem areas in soil conservation' (*circa* 1950). Land resource units (1,000's acres) were characterized by soil patterns (i.e. slope and erosion), climate, water resources, land use, and farming activities. Units were either continuous areas or several separate but nearby areas, and major land resource regions were distinguished according to state-wide agricultural use values and national planning initiatives.

Over a period of a few decades (1960-1980), the dominant perspective in land use planning and classification shifted away from an emphasis on agricultural, forest, and fish productivity (Ryder 1964, 1965) towards a more holistic perspective that considered multiple ecological factors. Robert Bailey of the USDA Forest Service was a key proponent of a more holistic approach. In 1983, he published a paper describing the 1976 version of an eco-regional delineation of the contiguous United States. This map and classification was based in part on earlier work by Crowley (1967) who developed a hierarchical classification with four levels (domains, divisions, provinces, and sections), which was classified on the basis of climatic similarity, climax plant communities and floristic composition. The technique used by Bailey was expanded to include the rest of North America and eventually the world (Bailey 1989). McNab et al. (1994) and Cleland et al. (1997, 2007) describe the most recent developments in (bioclimatic) eco-regional mapping throughout the conterminous U.S. by the Department of Agriculture, Forest Service. National Ecological Units in the U.S. are under a continual process of peer review, validation and refinement. Following Bailey's first delineation of USDA Forest Service sub-regions at the section level in 1976, a new national map of U.S. subsections was completed in 1994, and this map has since been updated (ECOMAP 2007, Cleland et al. 2007). The ecoregion concept has proven widely successful at the regional and state level as a spatial framework for natural resource management (USGS 1995).

The ECOMAP hierarchy includes ecoregions (10^4 to 10^6 square miles), subregions (10 to 10^3 square miles), landscapes (10^3 to 10^4 acres) and land unit ($<10^2$ to 10^3 acres) levels delineated at increasingly fine spatial scales (Cleland et al. 1997, ECOMAP 2007). Ecoregions contain global domains, continental divisions, and regional provinces that are delineated from the top-down using a process of expert review and GIS mapping. Domains around the globe are mapped according to broad climatic similarities in temperature, annual precipitation and evapo-transpiration, modified by topography as well as potential natural vegetation and biologically significant drainage systems. Continental divisions are distinguished mainly by water deficits (e.g. semi-arid, steppe, or arid desert) and winter temperatures. Regionally, provinces are recognized by continental weather patterns such as dry season length and duration of cold temperature extremes. Subregions contain sections and subsections, which are also delineated from the top-down by examining a combination of climate, geomorphic processes, topography and geology (strata). Landscapes are depicted by land-type associations that are characterized by similarities in geomorphic processes, geologic rock types, soil complexes, stream types, lakes, wetlands and vegetation associations. Soil complexes and plant communities are also used to delineate map units at this scale. Finally, land units (landtypes and landtype phases) are mapped in the field based on local topography, rock type and plant associations. (Cleland et al. 1997, ECOMAP 2007, McNab et al. 2007). Information collected from lower levels in the hierarchy (from subregions downwards) can be used to refine expert driven mapped boundaries from the bottom-up, creating a dual top-down classification.

In Ontario, Canada, the concept of forest ecosystem regions was developed by Hills (1961) and based on macroclimatic factors. Similar work was done in other parts of Canada (e.g. Krajina 1965, Crowley 1967). For example, the biogeoclimatic zones of British Columbia were described and delineated by Krajina (1965) and climatic regionalization is used in biophysical or ecological land classification throughout Canada (Wiken & Ironside 1977, Bailey 1983). Although the emphasis, as early as 1961, was on providing an ecological basis for natural resource management, it is clear from Hills' (1961) report that one of the main purposes of land classification was to subdivide the land into classes that reflected the relative utility of the land for humans in terms of agricultural crop capability classes and recommended use (e.g. forestry). Hill's system was hierarchical, including six spatially nested levels (site region, site district, landscape unit, landtype, site type, site phase). The system was fully adopted by Parks and Natural Heritage Section to help evaluate and design protected areas. But the classification was not fully adopted by forest and resource managers possibly because the system was too complex at the site and land-type identification levels (Uhlig & Baker 1994).

The Canadian Committee on Ecological Land Classification (CCELC) was formed in 1976 to develop the national ELC approach. Their approach was guided by earlier terrestrial ecological classifications (i.e. Hills et al. 1970, Wiken et al. 1986), as well as published national map sources such as the Ecoclimate Regions of Canada (Ecoregions Working Group 1989), the Physiographic Regions of Canada (Bostock 1970), the Soil Landscapes of Canada (Shields et al. 1991) and the Northern Land Use Information Series (Environment Canada and Indian and Northern Affairs Canada 1978-1986). The current ELC thus evolved out of earlier hierarchical classifications created by the CCELC and has seven levels of generalization: ecozone, ecoprovince, ecoregion, ecodistrict, ecosection, ecosites and ecoelements. Many of the variables selected to define levels in the National hierarchy correspond relatively well with the Hills et al. framework of 1970 (Uhlig & Baker 1994).

A total of 15 ecozones can be found across Canada, and these were delineated from the top-down according to broad physiographic and ecological similarities such as macroscale climate, human activity, vegetation, soils, geology and physiographic features. Within Canadian ecozones, there are 53 ecoprovinces, 194 ecoregions and over 1000 ecodistricts; ecodistricts are approximately linked to soil landscape polygons. The number and position of ecoprovinces across Canada reflects demand for a consistent spatial context for the purpose of broad conservation and resource purposes as well as the need to harmonize the delineation of ecoregional boundaries across the Canada-U.S. border (Marshall et al. 1999). Ecoregions are made up of unique landforms (or assemblages of regional landforms), meso-climate, vegetation, soil, water and regional human use patterns. Water is mentioned under the definition of ecoregions in terms of human activity patterns or 'water uses'. Water bodies are again mentioned under ecodistricts, which were characterized by distinctive assemblages of landform, relief, surficial geologic material, soil, *water bodies*, vegetation and land uses.

In 1997, a report by the NAFTA Commission for Environmental Cooperation recommended an integrated continental perspective across North America (Marshall et al. 1999). Robert Bailey also emphasized the need for information exchange and a common set of regional units in Canada and the U.S. when he compared the two countries' approaches to ecological regionalization in 1985. He found a lack of common terminology, different criteria and variations in the application of basic concepts underlying regionalization. For example, there were differences in the number of levels, naming conventions and philosophical differences in defining climax community (successional trends were emphasized in Canada whereas a single climax was emphasized in the U.S.). These differences make it difficult to compare the two systems directly. Nevertheless, both systems do share a common bioclimatic approach because they both integrate knowledge of climate, geology, geomorphology, soils and vegetation.

Since 1997, Marshall et al. (1999) list a number of new materials associated with the national ecological framework. The Provinces of B.C., Ontario, Saskatchewan, Manitoba and Nova Scotia have all produced ecoregional maps. But, there seems to have been relatively little activity on the national ELC since 1999, which was the end of funding from the Sustainable Forestry Initiative's five year term. However, Nature Conservancy Canada has made initiatives in the area of aquatic ecosystem classification (e.g. Blueprint project in Ontario and other projects in B.C.; Wichert et al. 2004, 2005).

The Province of Ontario's ELC is largely consistent with the Canadian National ELC (Fig. 2.1), but Ontario's system does not include 'ecoprovinces', and there are minor variations between provincial and national ecozone boundaries. These were made to ensure that the upper levels of the hierarchy were fully nested in Ontario's ELC, while both maintaining internal consistency at other scales and maximizing differences between adjacent categories. Ontario's ELC originally did not contain ecozones, nor did the classification system designed by Hills (1961), so this uppermost level was added later. Ecozones (3 in Ontario) can be interpreted as depicting the major bedrock regions in Ontario, which coincide with climatic and biotic differences (Crins & Uhlig 2000). At a broader scale, an ecozone represents a unique climatic biome at the global scale (Crins & Uhlig 2000). Ecoregions (14 in Ontario) are defined by additional climatic parameters, including precipitation, temperature (annual averages) and humidity (Crins & Uhlig 2000, Crins et al. 2009).

Ontario's ecodistricts were defined largely by surficial geology, bedrock, topography and parent soil materials. Currently, representation of Ontario parks is assessed at the ecodistrict scale. Within each of Ontario's 71 ecodistricts, the minimum guidelines are to represent at least 1% or 50 hectares of naturally occurring landform-vegetation (LV) associations. Additionally, park class targets for each ecoregion include; a wilderness park, a wilderness zone, a natural environment park and a waterway park or corridor along a waterway. Lower levels include ecosection, ecosite, ecoelement and plot levels, which are defined by variables including cover type (vegetation), soil moisture and texture. Most of the upper levels in the hierarchy were determined using semi-quantitative methods such as GIS spatial overlays of multiple variables, but boundary decisions were made by synthesizing expert opinion. A more rigorous and quantitative approach to ecoregionalization may be used in future (Crins & Uhlig 2000). Some of the lower levels were determined by multivariate statistical analysis (e.g. ordination).

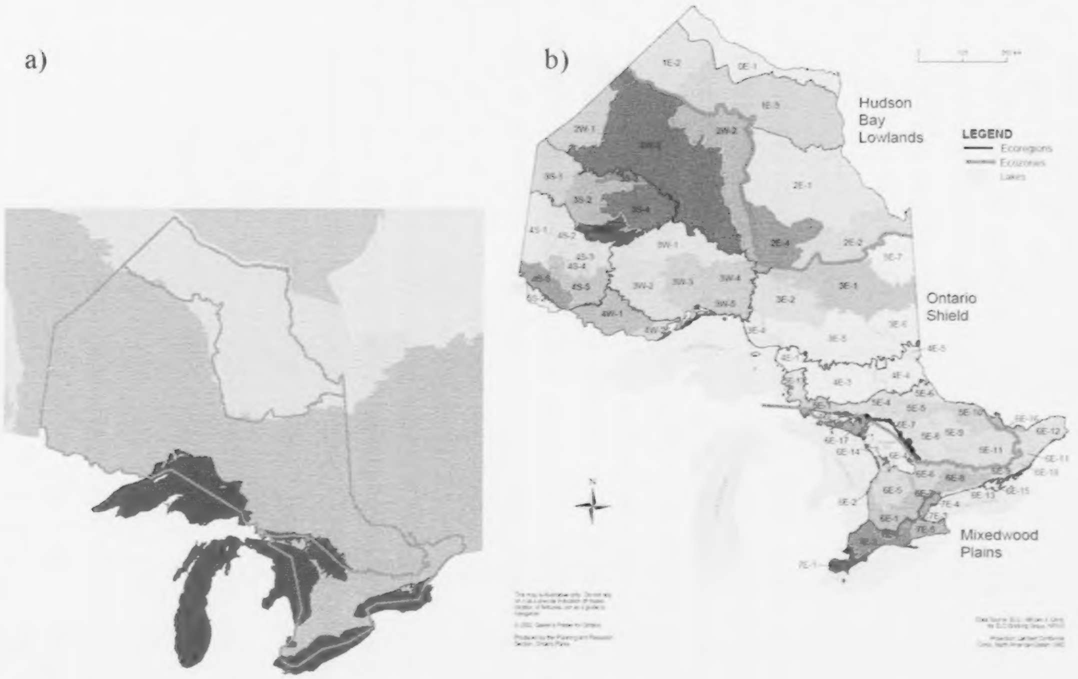

Figure 2.1. a) Comparison between Ontario terrestrial ecozone boundaries (*outlined in red*) and the Canadian National ecozone boundaries; b) Ecozones, ecoregions, and ecodistricts of Ontario.

2.2 Suitability of terrestrial classifications for aquatic ecosystems

The goal of classifying aquatic ecosystems stems out of this history of ecological land classification. It is possible to use terrestrial ecoregional classifications in order to assess potential diversity and process behaviour in aquatic systems. The theoretical basis for this linkage is the interconnected nature of land and water. Aquatic systems are integrated with their surrounding valleys and landscapes through the processes of runoff, sedimentation and the migration of biotic and chemical elements (Cleland et al. 1997).

Lotspeich & Platts (1982) provide a brief review of ecological land classification schemes in use at the time, noting that most do not attempt to integrate aquatic systems into their classification (including ELC classifications by Davis & Henderson 1976, Bailey 1976, Wertz & Arnold 1972). Lotspeich & Platts (1982) reasoned that *'land and water function together as a unit'* and separation of the two is artificial.

Omernik (1987) reviewed and evaluated two notable examples of land classifications in the U.S. for their ability to classify aquatic systems before developing a system of his own adapted specifically for aquatic environments. One of the classifications he reviewed was Austin's agricultural land use classification (1965). However, the scales at which map units were drawn and the obvious bias toward agricultural applications limited the applicability of this classification for defining ecoregions (Omernik 1987). The other was Bailey's 1976 map of ecoregions in the United States, which he found was inadequate for many stream areas because - at finer scales - ecosections were largely defined by potential natural vegetation, which did a poor job of capturing variation in stream systems.

Hawkins & Norris (2000) organized a symposium in 1998 designed to test the performance of regional terrestrial classifications in terms of their ability to partition the biotic variability among aquatic sites. Classification performance was assessed using Van Sickle's (1997) index of classification strength, which measures the difference between mean within-class and mean between-class similarity. Results of the symposium were presented in a series of articles published by the North American Benthological Society and synthesized by Hawkins et al. (2000). They identify the strengths and weaknesses of current approaches to (aquatic) classification. All terrestrial and aquatic classifications, according to Hawkins et al. (2000), can be divided into those based on *a priori* hypotheses and those based on *a posteriori* groupings of available inventory data (Gerritson et al. 2000). They reason that *a priori* classifications of sites are rarely based on a mechanistic, process-based understanding of the drivers behind biological productivity and diversity. The main findings of their initiative suggested that *'terrestrial landscape classifications will have limited application in aquatic bioassessments'*. In general classification strength was weak, although all landscape classifications examined were able to account for more variation in the aquatic biota (invertebrates, fish assemblages and diatoms) than would be expected by chance. In landscapes with strong gradients where the topography varied widely, terrestrial ecoregions were able to explain more of the variability in aquatic biota than where topography was more subtle. This result was consistent with several other studies reviewed by Hawkins et al. (2000).

Several reasons were posited to explain the weak relationships found: a lack of information on dispersal limitations over evolutionary time frames; the tendency for site similarity to decline with distance may account for more variability than measures of ecological processes; information on local habitat variability was lacking. They conclude their paper with two general recommendations for classifications. One, adopt a tiered approach with local habitat features forming the foundation of the classification and larger-scale features used for refinement. Two, be aware that classifications with good predictive ability may not necessarily aid understanding of the spatial and temporal stressors in the system.

In Ontario, Mandrak (1998) assessed the suitability of the then current terrestrial classification system in terms of its ability to predict aquatic biodiversity throughout the province. Consistent with findings summarized by Hawkins & Norris (2000), Mandrak (1998) found weak areal correspondence between terrestrial ecoregions and tertiary watersheds. Terrestrial regions corresponded poorly with fish faunal zones suggesting that the terrestrial classification system could not be used to inform conservation of freshwater biota (Wichert et al. 2004).

Chu & Jones (2010) went a step further and examined whether or not ecological land classifications of Ontario within the Great Lakes Basin of concern were able to capture major differences in stream temperature at eco-region and eco-district scales. Stream temperature is a critical aspect of ecosystem habitability for most species. They found that within class variation was greater than between class variation; so, for example, temperatures were more variable among sites within an eco-district class than between different eco-districts (Chu & Jones 2010).

This was not necessarily the case for water chemistry data. Earlier, Warry & Hanau (1993) found that land classifications developed in Canada by Rubec & Wicken (1983) and Wickware & Rubec (1989) showed good correspondence between groupings based on water chemistry and the existing land classification, though a more thorough statistical evaluation was considered necessary.

Robertson & Saad (2003) suggest several overall problems with using land-based ecoregional classifications to describe changes in water quality (Robertson & Saad 2003). Generally greater variations in water quality can occur within an ecoregion than among ecoregions possibly due to longitudinal zonation patterns (Hawkins et al. 2000) or the directional structure of these systems. Second, most ecoregions are defined using a variety of environmental characteristics, but the weighting of these characteristics varies in an unspecified way. So differences in water quality among ecoregions cannot be related to any particular set of environmental characteristics. Third, land-use is often an important characteristic used to delineate ecoregions, but land use is related to human activities, whereas baseline or reference water quality conditions are of interest.

Omernik & Bailey (1997) contend that ecoregions should not be evaluated in terms of their correspondence with the distributions of any particular community or resource (i.e. fish assemblages or macroinvertebrate distributions) because such comparisons often use an inappropriate hierarchical level of the ecoregional classification. They point out that terrestrial classifications are not designed to be the best framework for any one particular resource. Rather they are designed as research, assessment and management aids. Aquatic classifications likewise err by inferring that watersheds (or hydrologic units of some sort) can form the basis of dividing ecosystems. Watershed boundaries rarely correspond with characteristics related to distinct terrestrial environments (Omernik 1987). Xeric watersheds are not defined by the same processes as mesic and hydric watersheds, and in many areas (e.g. low relief areas) watersheds are simply difficult or impossible to define. Omernik & Bailey (1997) reasoned that in order to effectively extrapolate from watershed level data, one must know the larger ecological regions within which similar characteristics exist. Thus, both watershed-based approaches and ecoregion approaches to classification can be complementary.

2.3 The need for an aquatic classification system

In the Province of Ontario and around the world, land-based classifications are used extensively for protected areas planning (*Section 2.1*, Snelder et al. 2004a). Though many land-based ecoregional classifications account for significantly more variation in aquatic biota than random or null models, their ability to explain variation in aquatic community structure is generally low (Hawkins et al. 2000, Jenerette et al. 2002, Snelder et al. 2004a). As described above (*Section 2.1*), the poor fit could be related to a lack of information on both dispersal limitations over evolutionary time frames and changing local variability in aquatic environmental characteristics. In addition, ecoregions do not account for longitudinal patterns that are evident in most rivers; nor do they account for causal factors or aquatic process variables that lead to variation in aquatic biota at different spatial scales. Broad regions cannot account for fine-scale (within watershed) variability in temperature, topography, geology and land cover (Snelder et al. 2004a, Chu & Jones 2010). Currently in Ontario, ecodistrict boundaries do not respect watershed boundaries or water flow characteristics. Nor is there a generally accepted way to identify unique aquatic ecosystems at different spatial scales. The presence or absence of lakes in adjacent watersheds from the same ecoregion can lead to very different ecological characteristics (e.g. the hydrothermal regime) within similar aquatic systems (Jones 2010).

In April 2011, an OMNR survey was conducted on Aquatic Ecosystem Classification (AEC) to ascertain how various branches and sections within the Ministry could use an Aquatic Ecosystem Classification (Melles et al. 2011, Client Needs Report). A provincial AEC would help to identify unique aquatic ecosystems in Ontario at multiple spatial scales. Objectives of the survey were to: raise awareness; gather an aquatic classification science team; and document the needs of our internal OMNR clients. A section level response was requested and 17 sections from four divisions completed the survey. Comments were received from an additional five sections. The vast majority of respondents confirmed the utility of a Province-wide aquatic classification in relation to a wide variety of user-needs including: surveying and inventory (i.e. monitoring) of aquatic features and biota; identification of sites that may support species at risk; and strategic planning for resource use and development (e.g. renewable energy projects). Respondents identified a number of Provincial and Federal acts, policies, directions, and guidelines that would be supported by an aquatic classification (e.g. Ontario’s Biodiversity Strategy, Endangered Species Act, the Far North Act, Fisheries Act, and Watershed-based Fisheries Management Planning Guidelines).

In a relatively recent (2006) review of Ontario’s Protected Areas Legislation (i.e. the Provincial Parks and Conservation Reserves Act, PPCRA), a provision for an "aquatic" park class was added, pending development of a scientific and policy framework. One of the purposes of the PPCRA is to protect ecosystems representative of Ontario’s biodiversity. Section 8(8) of the PPCRA 2006 (which is yet to be proclaimed) includes the creation of this aquatic class of parks. An Aquatic Ecosystem Classification would help in the development and implementation of a scientific and policy framework for assessing representation of aquatic ecosystems in protected areas. An AEC would also help to identify gaps in protection and targets for securement for the new aquatic park class. Without an AEC, there is limited ability to assess how the Ontario PPCRA goals are being met with respect to protection of aquatic ecosystems in parks and conservation reserves, as well as with respect to how specific park class objectives are being achieved. An AEC would be useful for rationalizing the boundaries of protected areas and to inform management and restoration strategies at different scales. (Melles et al. 2011, Client Needs Report)

The client needs survey results concluded that Ontario needs an Aquatic Ecosystem Classification - now more than ever before - to support community-based, land-use planning in the Far North (Far North Science Advisory Panel 2010), aquatic parks planning, and to support species at risk and landscape-level, watershed-based planning in Southern Ontario.

Of course, any ensuing aquatic classification for Ontario would represent a hypothesis that needs to be tested with data (Snelder & Biggs 2002). An Ontario-wide ecological aquatic classification could be tested alongside the Provincial, land-based ELC and the Great Lakes region AEC for concordance with aquatic assemblage biodiversity and structure.

3 Aquatic classification approaches

The following section presents a quantitative assessment of some of the predominant approaches to aquatic ecosystem classification from around the world. Eighty-one classifications were selected for review (Table 3.1) with good representation from Canada (12), the United States (30), Europe (14) and global systems (12); only a few studies were from New Zealand (5), South Africa (3), Australia (2), and China (1). Ninety percent of the reviewed studies were published after 1980, and 79% were published after 1990. These studies represent a sample of the literature on aquatic classifications that have been conducted over the last several decades. Mere enumeration of the entire body of research done globally on aquatic classifications would require considerable space. However, some of the studies summarized here represent work in different years on similar classifications. We tried to avoid these as much as possible. However, different publications by the same research group often present extensions or substantial modifications to existing classifications, so they were given separate consideration.

Methods: As described in section 1.2, there are several different types of classification schemes. We determined whether selected classifications contained multiple hierarchical scales; whether the groupings made at each spatial scale were place-independent (e.g. geographic location is unimportant to classification of a site); place-dependent (e.g. delineated using spatially nested aqua-regions, aqua-districts, and aqua-sites); or some combination of the two; and we determined whether groupings were found using bottom-up statistical clustering and regression techniques, or top-down expert-driven approaches.

Variables used to create classifications were sorted into abiotic or biotic types and further subdivided according to their description (i.e. geoclimatic, geomorphic, hydrologic, chemical measures) or taxonomic focus group (e.g. fishes, benthic invertebrates, aquatic macrophytes, etc.). We noted whether classifications focused on catchment- or gradient-based (longitudinal) measures, and we noted whether measures of network position and connectivity were included. In addition, the smallest unit of classification was recorded.

Most papers on aquatic ecosystem classification begin by describing the purpose of their classification. Often there is more than one reason to classify freshwater systems, so a variety of purposes were recorded. Classifications may be descriptive and/or predictive and this is often stated explicitly. In addition, we recorded the criteria used for variable selection where indicated. Variables used to classify aquatic units were measured at a variety of spatial scales including: stream reach, river segment or segment catchments, whole system (lake, wetland, river and estuary), catchment, watershed, and/or ecoregion.

Results:

- The majority of classifications (64%) were multi-level classifications that included summaries at various spatial scales (Table 2). Almost half (47%) of all classifications examined took a regionalized approach. But of the classifications with more than one level, two-thirds took a regionalized (i.e. place-dependent) top-down and expert-driven approach. A combination of top-down and bottom-up approaches was used in almost one quarter of cases (23%), such that finer scales were classified in a place-independent manner, whereas larger scales were classified into spatially contiguous regions using expert driven approaches (Table 2). Place-independent approaches were used in all but one single-level classifications and in 40% of multi-level classifications.

- Research on effects of confluences and tributaries on stream characteristics is substantial, but there is less research on how lakes and wetlands might modify stream characteristics and vice versa. Only 27% of the aquatic classifications we reviewed included lakes in their classification; of these classifications, lakes were sometimes not explicitly classified; rather, lakes were examined in terms of how they modified the riverine environment.

Table 3.1 Descriptive details of global aquatic classification studies reviewed.

Index	Authors	Location	Topography	Lakes
1	Wolfe et al. (2007)	Alberta	lowlands	Yes
2	Ciruna et al. (2007)	British Columbia	coastal to mountainous (to 4660 m)	Yes
3	Miers (1994)	British Columbia	coastal to mountainous (to 4660 m)	Yes
4	Perrin & Blyth (1999)	British Columbia	coastal to mountainous (to 4660 m)	Yes
5	Mandrak (1999)	Ontario	lowlands, shield, arable (to 540 m)	Yes
6	Portt et al. (1989)	Ontario	lowlands, shield, arable (to 540 m)	Not explicitly
7	Ricker (1934)	Ontario	minor relief features	No
8	Wichert et al. (2004, 2005)	Ontario	minor relief features, moraines & ridges	Yes
9	Busch & Sly (1992)	Great lakes	minor relief features, moraines & ridges	Yes
10	Hudson et al. (1992)	Great lakes	minor relief features, moraines & ridges	No
11	Soranno et al. (2010)	US Great Lakes	US Great Lakes area and beyond	Yes
12	Kim & Hawkins (1993)	Northeastern USA	coastal to mountains (1500 m)	No
13	Weitzel et al. (2003)	Central US	mostly flat, rolling hills	Yes
14	Lotspeich & Platts (1982)	Continental US	coast to coast (US)	Yes
15	Maxwell et al. (1995)	USA	coast to coast (US) to coast	Yes
16	Cowardin et al. (1979)	USA	coast to coast (US)	Yes
17	Canfield et al. (1983)	Florida	gently rolling, plateau, (up to 105 m)	Yes
18	Peterson et al. (2009)	SW Georgia	coastal plains to mountains (1459 m)	No
19	Savage & Rabe (1979)	Idaho	mountainous (3862 m)	No
20	Kuehne (1962)	Kentucky	relatively flat (365-457 m)	No
21	Brenden et al. (2008)	Michigan	generally level (174-603 m)	No
22	Seelbach et al. (2006)	Michigan	generally level (174-603 m)	No
23	MIDNR (1967)	Michigan	generally level (174-603 m)	No
24	Sowa et al. (2007)	Missouri	flat prairie to Ozark plateau (540 m)	No
25	Cupp (1988)	Washington	coastal to mountains (to 4392 m)	No
26	Besaw et al. (2009)	Vermont	valleys & mountains (1293 m)	No
27	Thompson et al. (2006)	SE Australia	mountainous (ave 1000 m)	No
28	Wimmer et al. (2000)	Austria	rolling hills to mountains (to 3798 m)	No
29	Heiner et al. (2011)	China, Yangtze	Tibetan plataeu & mountains (to 5000 m)	No
30	Hering et al. (2006)	Europe	sites (to 800 m)	No
31	Noble et al. (2007b)	Europe	EU, coastal to mountains (to 5000 m)	No
32	Lee et al. (1997)	Korea	mostly hilly & mountains (to 1950 m)	No
33	Biggs et al. (1990)	New Zealand	coastal to mountainous (to 3754 m)	Not explicitly
34	Leathwick et al. (2010, 2011)	New Zealand	coastal to mountainous (to 3754 m)	Not explicitly
35	Snelder et al. (2003, 2004)	New Zealand	coastal to mountainous (to 3754 m)	Yes
36	Snelder et al. (2007)	New Zealand	coastal to mountainous (to 3754 m)	No
37	Kleynhans et al. (2005)	South Africa	coastal to inland mountainous (to 3408 m)	Not explicitly
38	Rivers-Moore & Goodman (2010)	South Africa	coastal to inland mountainous (to 3408 m)	No
39	Wadeson (1999)	South Africa	coastal to inland mountainous (to 3408 m)	Not explicitly
40	Poquet et al. (2009)	Spain	lowlands to mountains (to 3482 m)	No
41	Ahl (1994)	Sweden	flat, gently rolling, mountains in W (to 2111 m)	Yes
42	Harris et al. (2000)	United Kingdom	rugged hills & low mountains (to 1343 m)	No
43	Hall & Minns (1999)	United Kingdom	rugged hills & low mountains (to 1343 m)	Not explicitly

Index	Authors	Location	Topography	Lakes
44	Holmes et al. (1998)	United Kingdom	rugged hills & low mountains (to 1343 m)	No
45	Wright et al. (1984, 1998), Clarke et al. (2003)	United Kingdom	rugged hills & low mountains (to 1343 m)	No
46	Dodds et al. (1998)	Global	varied	No
47	Frissell et al. (1986)	Global	varied	Yes
48	Hakanson et al. (2007)	Global	varied	Not explicitly
49	Hawkins et al. (1993)	Global	varied	No
50	Montgomery & Buffington (1993, 1998)	Global	varied	No
51	Nanson & Croke (1992)	Global	varied	No
52	Nanson & Knighton (1996)	Global	varied	No
53	Pennak (1971)	Global	varied	No
54	Rosgen (1994, 1996)	Global	US / global	No
55	Whiting & Bradley (1993)	Washington	coastal to mountainous (to 4392 m)	No
56	Higgins et al. (2005)	USA	coast to coast (US) to coast	Yes
57	Dollar et al. (2007)	Global	varied	Not explicitly
58	Brierley & Fryirs (2000), 'Thomson et al. (2004)	Australia	mostly low plateau deserts (to 2229 m)	No
59	Devito et al. (2005)	Alberta	Boreal forest	Not explicitly
60	Robertson & Saad (2003)	Midwest USA	flat to rolling hills	Not explicitly
61	Brown et al. (2003, 2009)	France	alpine areas (to 3144 m)	No
62	Cohen et al. (1998)	France	N & W mostly flat, S&E mountains (to 4807 m)	No
63	Wasson et al. (2002)	France	N & W mostly flat, S&E mountains (to 4807 m)	Yes
64	Molnar et al. (2002)	Global	varied	No
65	McCarthy et al. (2006)	Labrador	coastal to mountains (to 1652 m)	Yes
66	Craig & McCart (1975)	Alaska & Yukon	coastal, tundra to mountains (to 6198 m)	No
67	Hawkes et al. (1986)	Kansas	plains to hills (207-1232 m)	Not explicitly
68	Hughes et al. (1988)	Oregon	coastal to plateaus & mountains (to 2700 m)	Not explicity
69	Sable & Wohl (2006)	Oregon	coastal to plateaus & mountains (to 2700 m)	no
70	Mock (1971)	Pennsylvania & New Jersey	rolling hills & lowlands (to 980 m)	No
71	Jensen et al. (2001)	Northwestern USA	mountains (to 4392 m)	Yes
72	Hughes et al. (1986)	USA	coast to coast (US)	Not explicitly
73	Montgomery (1999)	USA	coast to coast (US)	No
74	Parrott et al. (1989)	USA	coast to coast (US)	No
75	Harding & Winterbourn (1997)	New Zealand	rough coastline, plains, mountains (to 3754 m)	No
76	Calle-Martinez & Casas (2006)	Spain	mountainous (to 3482 m)	No
77	Johnson (1999)	Sweden	flat, gently rolling, mountains in W (to 2111 m)	Yes
78	Verdonschot (2000)	Global	varied	No
79	Omernik (1987)	USA	coast to coast (US)	No
80	Bryce & Clarke (1996)	Oregon	coastal to plateaus & mountains (to 2700 m)	No
81	Abell et al. (2008)	Global	varied	Yes
			Percent of total	**27.16%**

Table 3.2 Classification approach and number of hierarchical levels in global AECs.

Index	Authors	Place independent	Place dependent (regions)	Top-down	Bottom-up	Number levels	Smallest unit of classification
1	Wolfe et al. (2007)	X			X	1	lake basin areas or catchments (ave 68 km^2)
2	Ciruna et al. (2007)		X	X	X	3	whole river and lake primary watersheds
3	Miers (1994)		X		X	7	segments, whole lakes, wetlands, marine & estuary
4	Perrin & Blyth (1999)		X	X	X	3	watershed 'group' - catchment based
5	Mandrak (1999)		X		X	3	tertirary watersheds
6	Portt et al. (1989)		X	X		3+	watersheds
7	Ricker (1934)	X				3	hydrologic units
8	Wichert et al. (2004, 2005)	X	X	X		5	coastal areas, streams, wetlands, inland lakes
9	Busch & Sly (1992)	X				5	wetlands, lakes, estuaries, and rivers
10	Hudson et al. (1992)	X	X		X	3+	stream segments
11	Soranno et al. (2010)		X		X	1+	lakes used as an example of their system
12	Kim & Hawkins (1993)	X			X	1	gauged streams
13	Weitzel et al. (2003)		X	X	X	4	stream segment
14	Lotspeich & Platts (1982)		X	X		5	watersheds (i.e., catchments around 1st order streams)
15	Maxwell et al. (1995)		X	X		7	hydrologic units
16	Cowardin et al. (1979)	X		X		5	wetlands, lakes, marine areas, estuaries, and rivers
17	Canfield et al. (1983)	X				1	lakes
18	Peterson et al. (2009)	X				1	stream segment
19	Savage & Rabe (1979)	X			X	1	stream segment
20	Kuehne (1962)	X		X		1	stream segment
21	Brenden et al. (2008)	X			X	2	river valley segments (10's km)
22	Seelbach et al. (2006)	X		X	X	1	river valley segments (10's km)
23	MIDNR (1967)	X				3	stream segments (10's km)
24	Sowa et al. (2007)		X	X	X	8	river valley segments (10's km)
25	Cupp (1988)	X		X		1	river valley segments (10's km)
26	Besaw et al. (2009)	X		X	X	1	reaches with drainage areas (ave. 90 km^2)
27	Thompson et al. (2006)	X		X		1	stream reaches between breaks in channel slope
28	Wimmer et al. (2000)		X	X		2	catchments > 10 km2
29	Heiner et al. (2011)		X	X		3	catchments around similar stream segment networks
30	Hering et al. (2006)	X	X	X		1	stream sites, small/mid-sized (catchments <1000 km^2)
31	Noble et al. (2007b)	X		X	X	7	fish species assemblages
32	Lee et al. (1997)	X			X	1	stream (reference site) water quality & fish species
33	Biggs et al. (1990)		X	X		3	catchments (ave 406 km^2)
34	Leathwick et al. (2010, 2011)	X			X	1	stream segments between adjacent confluences
35	Snelder et al. (2003, 2004)	X		X	X	6	river valley segments
36	Snelder et al. (2007)	X			X	1+	stream segments
37	Kleynhans et al. (2005)		X	X		2-4	segment catchments
38	Rivers-Moore & Goodman (2010)		X	X	X	3	quaternary (4th order) catchments
39	Wadeson (1999)		X	X		6	morphological unit & associated hydraulic biotopes
40	Poquet et al. (2009)	X			X	1	stream reference sites
41	Ahl (1994)	X			X	1	small gauged reference watersheds
42	Harris et al. (2000)	X			X	1	river discharge and temperature regimes
43	Hall & Minns (1999)	X			X	1	catchment
44	Holmes et al. (1998)	X			X	3	1 km stream reaches
45	Wright et al. (1984, 1998), Clarke et al.	X			X	1	multiple longitudinal reference sites (river reaches)
46	Dodds et al. (1998)	X			X	1	stream sites within low order to large rivers
47	Frissell et al. (1986)		X	X		5	watersheds divided down to microhabitat classes
48	Hakanson et al. (2007)	X			X	1	bioindicators (e.g., chl) coastal, transition or deep water
49	Hawkins et al. (1993)	X		X		3	channel geomorphic units (e.g., homogeneous depth/flow)
50	Montgomery & Buffington (1993, 1998)		X	X		5	channel units (pool/riffles), reaches/segments/watersheds
51	Nanson & Croke (1992)	X	X	X		3	floodplains
52	Nanson & Knighton (1996)	X		X		2	anabranching rivers
53	Pennak (1971)	X				1	stream segments
54	Rosgen (1994, 1996)	X				4	stream reaches
55	Whiting & Bradley (1993)	X		X		3	headwater of stream
56	Higgins et al. (2005)		X	X	X	4	river valley segments and small to med. lakes
57	Dollar et al. (2007)		X	X	X	5	river ecosystems
58	Brierley & Fryirs (2000), Thomson et al.		X	X		4	catchment
59	Devito et al. (2005)	X		X		1	hydrologic response units (catchments)
60	Robertson & Saad (2003)	X			X	3	stream catchment
61	Brown et al. (2003, 2009)	X		X		1	stream reach
62	Cohen et al. (1998)		X	X		3	stream reach
63	Wasson et al. (2002)		X	X		2	catchment area
64	Molnar et al. (2002)	X		X	X	4	catchment
65	McCarthy et al. (2006)	X		X		3	river segments, lakes (littoral & profundal zones)
66	Craig & McCart (1975)	X		X		1	entire length of stream
67	Hawkes et al. (1986)	X			X	1	fish assemblage ecoregions
68	Hughes et al. (1988)	X	X		X	1	ichthyogeographic regions
69	Sable & Wohl (2006)	X			X	1	stream reaches in 4th & 5th order stream segments
70	Mock (1971)	X		X		2	channel link (e.g., bifurcating, tributary, source)
71	Jensen et al. (2001)	X	X	X	X	6	subwatersheds (10-100 km^2)
72	Hughes et al. (1986)		X	X		3	reference sites
73	Montgomery (1999)		X	X		5	process unit
74	Parrott et al. ([illegible])		X	X		4	hydraulic unit
75	Harding & Winterbourn (1997)		X	X		2	stream catchments
76	Calle-Martinez & Casas (2006)	X			X	1	stream site
77	Johnson (1999)		X	X		2	lake
78	Verdonschot (2000)		X	X	X	5	microhabitat
79	Omernik (1987)		X	X		1	ecoregion
80	Bryce & Clarke (1996)		X	X		5	landscape level ecoregion
81	Abell et al. (2008)		X	X	X	1+	ecoregion
	Percent of total	60.49%	46.91%	61.73%	48.15%	64.20%	

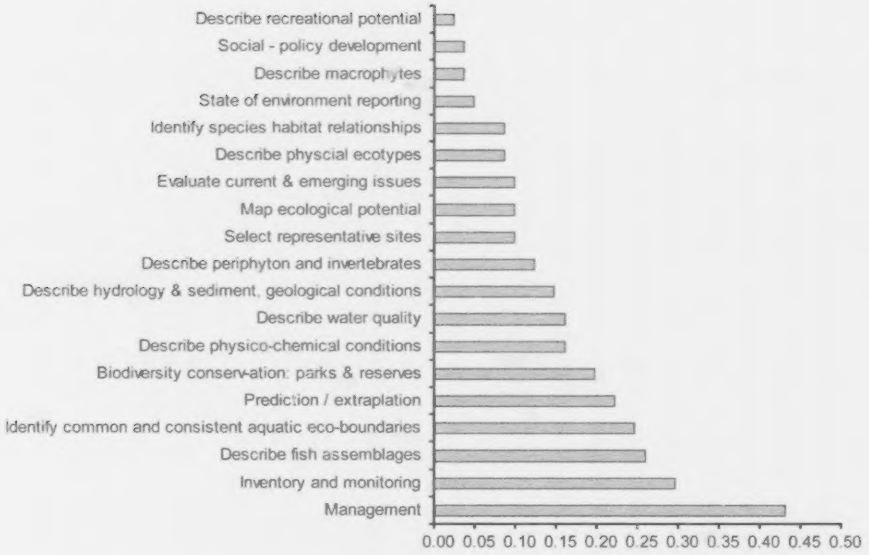

Figure 3.1. Proportion of studies with the stated purpose(s) for aquatic ecosystem classification, based on our review of $n = 81$ classifications.

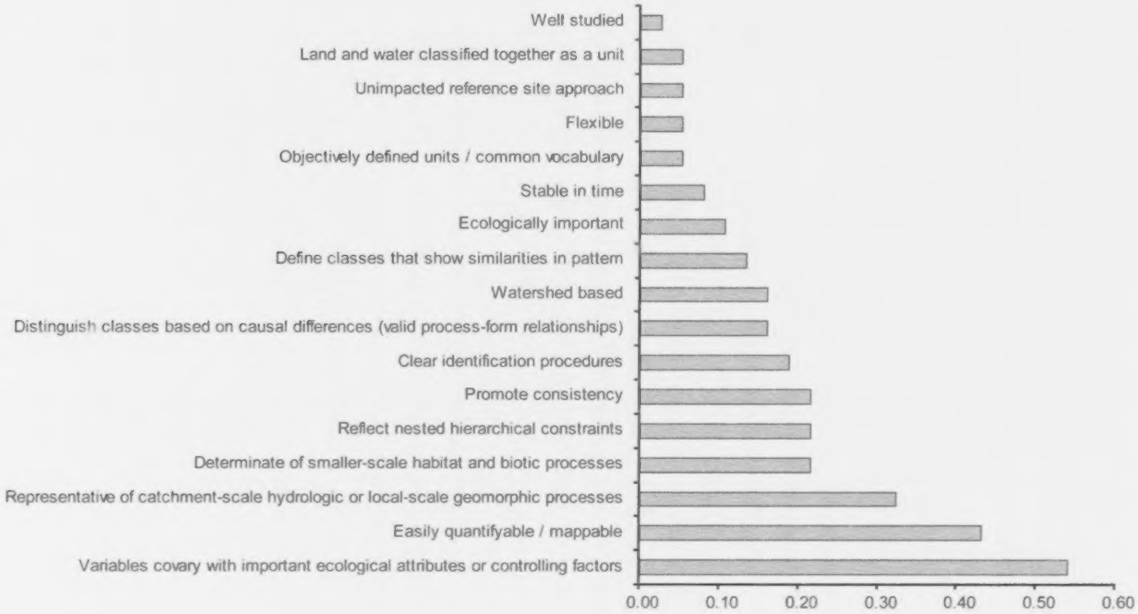

Figure 3.2. Proportion of studies that listed these criteria for variable selection ($n = 37$).

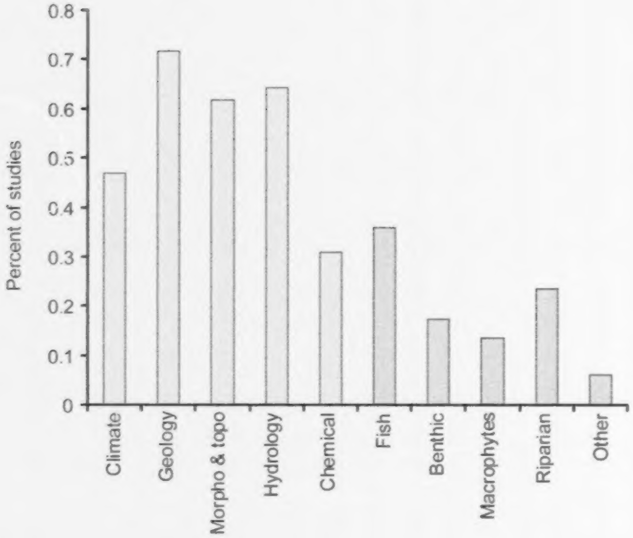

Figure 3.3. Proportion of studies that used these classes of dominant variables in their classifications ($n = 81$). Abiotic variables are shown in light orange and biotic variables are shown in green.

- No studies truly integrated wetlands and groundwater systems to create a cohesive aquatic classification. That is, wetland and known groundwater aquifers were not explicitly classified, nor were such classes incorporated into an aquatic ecosystem classification.

- Management as well as inventory and monitoring were the most commonly stated reasons for developing an aquatic ecosystem classification (Fig. 1). A description of fish assemblages was important in approximately 25% of the studies, and identification of candidate aquatic parks or reserves was a focal consideration in approximately 20% of the studies (Fig. 1).

- Of the studies that listed criteria for variable selection, the majority selected variables that co-varied with key ecological factors or controlling processes. The availability of information that was easily quantified was an important consideration (Fig. 2).

- In the past, many European and North American classifications were based on dominant fish species assemblages arranged longitudinally into fish community zones (e.g. Carpenter 1928, Anonymous 1967). Though biotic factors are still prevalent in many classifications developed since 1980, Figure 3.3 shows that abiotic factors are much more prevalent in the classifications of today. For example, less than 40% of the classifications we reviewed used data on fish assemblages to inform aquatic classes. The four most prevalent abiotic factors measured or modelled included geology (70% of studies), hydrology (~60%), morphology/topology (~60%), and climate (~48%).

- A wide range of variables have been used to assess aquatic ecosystem diversity at different spatial scales (Fig. 4). Variables measured at the level of river reaches can be assessed at much finer spatio-temporal scales (e.g. current speeds, dominant bed materials, river width:depth ratios) than variables summarized at the watershed level (e.g. physiography, geologic history). The segment level was the focus of many classifications and often formed the smallest unit of classification. A majority of variables were assessed at the segment level.

- Variables that account for the position of an aquatic feature within a watershed or basin (e.g. stream order, drainage area, network position) were generally included at a variety of scales.

- Few variables were integrated across different scales. Most classifications attempted to capture different processes at each scale.

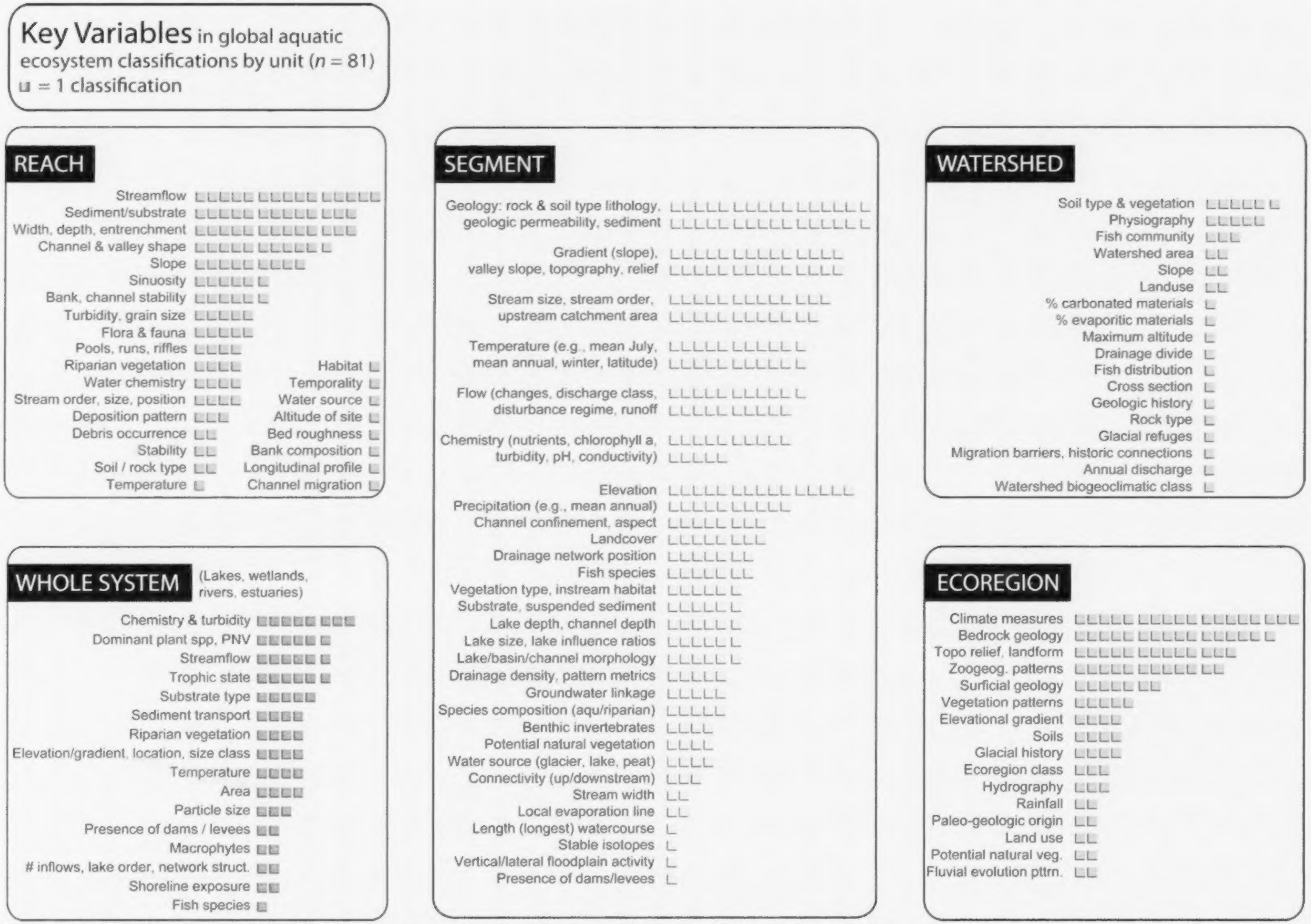

Figure 3.4. Key variables used in global aquatic ecosystem classifications. PNV: potential natural vegetation.

3.1 Early classifications and global classifications

Early European and North American stream classifications were biotic and based on dominant game fish species (e.g. Carpenter 1928, Anonymous 1967). Sections of stream were known as trout, grayling, barbell, and bream zones ordered sequentially along the stream in longitudinal zones (Hawkes 1975). For example, Illies (1961) writing in German, recognized eight zones within any drainage system ranging from springs to lowland rivers and the river mouth (Pennak 1971). This zonation system was generally accompanied by thorough faunistic studies that characterized the communities within each zone.

Pennak (1971) was among the first to suggest that classification of running waters exclude faunal information. He suggested that biotic criteria (or biological indicator species) are of little value because of the environmental plasticity of various species: some species can be found in a wide variety of habitats around the world and thus provide a poor indication of stream characteristics. According to Pennak, only stream segments or (preferably) stream habitats may be properly compared and contrasted from place to place because rivers vary markedly both longitudinally and from tributary to tributary. He suggested 13 abiotic (physico-chemical) variables be used, including aspects of stream hydrology (flow, current speed), geology (substrate), morphology (stream width), temperature, chemistry and vegetation (i.e. the presence and relative abundance of rooted aquatics and stream-side vegetation like brush and woodland), though these last two variables can actually be considered biotic. Of all these variables, substrate has perhaps the greatest potential biological significance (Pennak 1971). Pennak inferred that when these 13 river characteristics are compared alongside assessments of periphyton and bottom fauna, the classification becomes particularly meaningful. However, his system of world-wide stream classification was not widely adopted.

Hynes (1975) suggested that early attempts to understand how streams differed focused on local phenomena at the field-scale. These stream characteristics were easily related to the biotic community of interest (primarily fishes), which led to classifications of watercourses with millions of potential stream types (e.g. Pennak's 1971 system had more than 92 million possible categories). Later, the complexity of river and stream ecosystems was addressed using hierarchical, multi-scaled approaches, and Frissell et al. (1986) established a framework for global stream classifications based on hierarchy theory. In this framework, the entire stream system responds to gradual processes that operate over large spatial scales (e.g. glaciation, annual sedimentation). Whereas stream segments, reaches, and river features such as pools or riffles respond to processes operating over shorter and shorter temporal periods at finer and finer spatial scales.

Classifications based hierarchy theory assume that larger-scale and longer-term processes set constraints on shorter-term systems operating at smaller and smaller scales (Frissell et al., 1986). This assumption is generally reflected in the spatially nested hierarchical framework of land-based classifications. Indeed, hierarchy theory is really only pertinent to nested hierarchies where upper levels completely contain the next lower levels (Valentine & May 1996, Wu & David 2002). Furthermore, hierarchy theory assumes that the relationship between upper and lower levels is generally asymmetric: upper levels (long-term, large-scale dynamics) exert constraints on lower levels (short-term, small scale dynamics), and lower levels provide initial conditions for upper levels, but small scale dynamics may not have the same impact on the system as a whole (Frissell et al., 1986; Wu & David, 2002). Any event that causes shifts in a large-scale system will change the capacity of all lower-level systems it encompasses, but events that affect smaller-scale habitat characteristics may not affect larger-scale system characteristics (Frissell et al. 1986, Naiman et al. 1992, but see Melles et al., 2012 for a different view).

Frissell et al.'s (1986) hierarchical approach to stream classification has been modified, and evaluated by various authors with applications at global and continental scales (e.g. Montgomery and Buffington 1993, 1998; Hawkins et al. 1993; additional references can be found in review by Melles et al. 2012). More recent authors, notably Dollar et al. (2007), have stretched these ideas a step further by suggesting a framework that presents parallel organizational hierarchies in geomorphology (from geomorphic provinces and drainage basins to individual particles of sediment), hydrology (from volume and discharge to turbulence) and ecology (from landscape and ecosystem to species and organism).

Hawkins et al. (1993) proposed a global classification system that separates all stream reaches into slow and fast moving sections. These sections are then further classified as scour or dammed pools, turbulent or non-turbulent waters. At their most detailed level for example, turbulent waters are classified as falls, cascades, rapids, riffles and chutes. Though the authors describe their classification as hierarchical, their approach differs substantially from the spatially-nested hierarchical approach proposed by Frissell et al. (1986). Hawkins et al.'s (1993) classification does not group sections of stream of equal rank together to form upper levels in the hierarchy (as per Frissell et al. 1986). What Hawkin's et al. suggest is that lower levels descend from upper levels, rather like phylogenetic or taxonomic trees where daughters descend from mothers who are descendents of grand-mothers (Valentine and May 1996). In order to be classified as a riffle in their system, a section of stream must first be classified as fast moving and then turbulent. These different approaches to hierarchical classification add to the confusion over what can be inferred from hierarchical groupings (Melles et al. 2012). Hawkin's et al. (1993) suggest that their approach can be easily integrated into the spatial hierarchical system proposed by Frissell et al. (1986) based on hierarchy theory.

Other approaches to global river classification can be found. Nanson and Croke (1992) describe a classification of global river floodplains into thirteen classes based on stream power and sediment characteristics. Whereas Dodds et al. (1998) suggested a global classification system that grouped streams according to their trophic state. Welcomme et al. (1989) made an earlier attempt to promote a global river morphoedaphic index model (RMEI) based on variables like length of the river channel, drainage basin area and nutrient surrogates (e.g. total dissolved solids and conductivity).

In the 1980's and 1990's, Rosgen (1985, 1994, 1996) classified streams using a geomorphic approach into a potential 416,000 abiotic classes according to a variety of morphological characteristics (e.g. gradient, channel sinuosity, substrate). There are four hierarchical (place-independent) levels in Rosgen's system with each level more specific than the next. Level I characterizes rivers based on broad geomorphology into eight channel types, A through G, which indicate decreasing channel slope. Stream type differs from stream order because Rosgen judged stream order to be a misleading measure of stream size due to differences among hydro-physiographic provinces, geologic characteristics and the scale of the maps used to determine stream order. Level II characterizes rivers based on six types of dominant bed material (e.g. sand, gravel, bedrock). Level III describes the existing condition or "state" of the stream in terms of stability, response potential and function, whereas level IV is proposed as a verification level in which detailed measurements of process relationships can be obtained. The basic premise of Rosgen's work on river classification is that channel morphology is governed by the laws of physics '*through observable stream channel features and related fluvial processes.*' Rosgen arrived at classification from the perspective of physical engineering, and his system of stream classification is still widely used and taught among (physical/hydraulic) engineers. The advantage of Rosgen's system is that most of the data can be acquired from topographic maps and aerial photos (Portt et al. 1989), but some of the classification criteria in his system are somewhat subjective (e.g. 'moderately entrenched and moderately confined' as opposed to 'moderately entrenched and well confined' streams).

Rosgen's work has been criticized for its' lack of consideration for surrounding catchment characteristics, which are functionally linked to the structure and dynamics of streams (Hynes 1975, Gordon et al. 2004). Rosgen's system has been further criticized by various authors that argue stream types are irrelevant in the face of primary variables that shape the channel such as grain size, slope and discharge (Montgomery and Buffington 1997, Gordon et al. 2004). Grain size is emphasized only at level two in Rosgen's system and discharge is not considered at all. Hynes fervently opposed the large number of possible stream types identified using this kind of abiotic classification writing that they are *'little more useful than listing* streams *by name'* (Hynes 1975). Such abiotic systems require continuous monitoring and are therefore largely impractical. Some do not consider temperature, while others do not relate the division of variables to biotic communities (e.g. Pennak 1971).

Another geomorphic approach to (mountain stream) channel classification was proposed by Montgomery and Buffington (1997) who summarized a theory for the origin of distinct channel types with broad applicability around the globe. They recognized a general downstream progression of distinct reach types in the following order: cascades, step-pools, plane-bed or forced pool-riffle, and pool riffle. Additionally, they distinguished two more reach types: bedrock and dune-ripple.

- **Cascades** of highly turbulent flow over lots of cobbles and boulders generally occur over steep slopes within confined valley walls
- **Step-pools** are characterized by longitudinal steps formed by the accumulation of large rock fragments with more tranquil flows through intervening pools
- **Plane-bed channels** lack discrete bars or steps; they have low width to depth ratios; they occur at high to moderate slopes in relatively straight channels; and they have relatively even flow over cobble and gravel beds.
- **Pool-riffle channels** form out of a series of undulating high and low points or depressions: they typically form in relation to each other. Pool-riffle channels typically occur at moderate to low gradients; they occur in unconfined valleys

  with established floodplains; and they have substrate sizes that vary from sand to gravel and cobble.
- **Dune-ripple channels** typically occur at low gradients in sand-bed channels and are distinguished by washboard like sequences of ripples and or dunes.
- **Colluvial channels** are small headwater streams that flow over relatively low grade slopes and deposits of colluvial fill from the surrounding hill slopes. They are characterized by shallow and ephemeral flow.
- **Bedrock channels** lack an alluvial bed so water flows directly over bedrock. They are generally confined by valley walls, have high transport capacity, steep channel gradients, and/or deep flows.

Beyond describing these channel features or channel types, Montgomery and Buffington (1997) couple these reach morphologies with in-stream channel processes, hill slope processes and external processes such as channel confinements, riparian vegetation and woody debris. They suggest that channel features reflect differences in bed material size, roughness, and configuration, which are the result of differences in transport capacity and sediment supply from surrounding hill slopes, upstream processes, and/or riparian processes and disturbances. The ratio of transport capacity to supply differs between the different channel types and they can be further aggregated into source, transport, and response segments at watershed scales. Though the reach morphologies described by Montgomery and Buffington (1997) generally pertain to fine resolutions and small-scale features.

Recently, Abell et al. (2008) described the world's first biogeographic regionalization of freshwater ecosystems based on the distribution and composition of freshwater fish species. Fish species occurrence patterns are a reflection of processes that led to species dispersal, survival and/or isolation over large extents and long time frames (e.g. mountain building, speciation, glaciation; Tonn 1990). Species occurrence patterns also reflect processes that occur over smaller (regional) extents and shorter time frames (e.g. climatic and physiographic patterns, regional watershed barriers, etc.; Tonn 1990). Abell et al. (2008) map unique freshwater ecoregions based on distinct freshwater fish communities and fish species assemblages, which are subject to these regional and large scale processes. They qualify their classification by indicating that their freshwater ecoregional boundaries do not necessarily reflect species turnover or range edges. Rather they are meant to describe broad patterns in species composition and associated ecological and evolutionary processes (Abell et al. 2008).

Earlier work by Abell et al. (2000) mapped the freshwater ecoregions of North America on the basis of work by Maxwell et al. (1995). In fact, the 2008 global classification adopts Maxwell et al.'s U.S. subregions (i.e. major drainage systems defined by endemism and unique fish communities) with relatively minor modification. As Maxwell et al. (1995) used the principles of hierarchy theory as the scientific basis for their classification, so too is Abell et al.'s (2008) classification based on the principle that each component in the hierarchy is a discrete functional entity that is part of a larger whole. Processes operating at larger scales are assumed [illegible] set constraints for smaller scale systems that are completely and equivalently nested within them. For example, secondary watersheds are assumed to be constrained by the zoogeographic setting of primary watersheds delineated on the basis of geomorphic features that reflect glacial history. The Canadian ecoregional delineation in Abell et al's (2008) classification was based on separate cluster analysis of fish occurrence data in secondary watersheds within each of nine primary watersheds in Canada.

Several conservation applications have been suggested for Abell et al.'s new aquatic ecoregional map and associated species data including the use of this map to help evaluate the uniqueness of wetlands nominated under the Ramsar Convention on Wetlands (which includes all freshwater systems, Abell et al. 2008). The map may provide a framework to identify broad scale gaps in global protection and conservation efforts assuming that fishes serve as proxies for the distinctiveness of biotic assemblages at ecoregional scales (Abell et al. 2008). However, the authors do recognize that analyses of other taxonomic groups (i.e. freshwater invertebrates) would likely reveal different patterns, because of differences in animal dispersal abilities. They also recognized that their results are scale dependent. Therefore, it is difficult to reconcile how conservation plans designed to target biodiversity can be met by these heterogeneous freshwater ecoregions with boundaries that relate to constraints on fish zoogeography irrespective of the distribution of aquatic invertebrates, mussels, aquatic macrophytes, reptiles, or amphibians. Additionally, freshwater ecoregions encompass complex aquatic networks with longitudinal directionality. Species turnover within these ecoregions is typical: they are not homogeneous units in terms of the species they aim to describe.

3.2 Classifications in Canada

The first stream classification system in Ontario was created by William Ricker in 1934 for some southern Ontario streams, but the methodology was generally applicable in other regions, and the approach has been widely cited (Pennak 1971). Ricker developed his stream classification system based on investigations of the Mad River in Grey and Simcoe counties, Southern Ontario. The system was based predominantly on sport fishes and their habitat (e.g. Salmoninae). Rivers were classified according to their size, the physical and chemical properties of the water, the texture of the river substrate and the abundance and diversity of river flora and fauna. The following main groupings were made: creeks could be either spring or drainage, and rivers were classified as slow or swift trout streams and warm rivers. Portt et al. (1989) suggest that because Ricker's classification included physico-chemical characteristics, it went one step beyond a simple biotic classification. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has developed National Freshwater Biogeographic Zones that are used for identifying designatable units of freshwater fishes and molluscs. These zones are meant to reflect differences in evolutionary history.

In the following subsection, we expand on a few of the previous efforts towards aquatic ecosystem classification in Ontario described in background section 1.1: notably we examine AEC frameworks outlined by Portt et al. (1989), Mandrak (1999), and the Blueprint project AEC.

3.2.1 Ontario

Portt et al. (1989) As mentioned in section 1.3, these authors recommended a geoclimatic classification system be implemented in Ontario because climate and geology are critically important drivers of species' habitat and dispersal (Lotspeich and Platts 1982, Portt et al. 1989), and geoclimatic classifications recognize the direct and indirect linkages between land and water. Additionally, geoclimatic classifications are based on abiotic data that are more readily available and inexpensive to obtain. As opposed to biotic or physical/chemical classifications, geoclimatic classifications do not require detailed knowledge of habitat characteristics or biota, which would be difficult if not impossible to obtain throughout Ontario. Portt et al. recommend that physiographic units be used (e.g. kame moraines, clay plains) as the fundamental units of classification because physiography is more constant through time than climate and it is easier to assess the relationship between physiography and stream habitats than between climatic variables and stream habitats.

To begin with, Portt et al. (1989) recommend developing models of the relationships between physiography, stream habitat and biota within smaller selected areas of southern Ontario, not heavily influenced by human activities (i.e. urbanization, dams, channelization). For biota, they recommend compiling information on fishes and possibly invertebrate species distributions as well as their habitat characteristics. Stream order, a reflection of stream size, is a recommended attribute for classification because it integrates a variety of physical parameters such as mean stream width and depth, gradient, altitude, and in many cases, fish species richness and diversity (Barila et al. 1981, McGarvey & Milton Ward 2008, McGarvey 2010). However, Portt et al. were quick to point out that stream order is not the primary factor determining stream fish communities in North America (*citing* Lotrich 1973, Evans and Noble 1979, Mathews 1986 and Mahon et al. 1979). To evaluate the utility of physiography as the basis for stream classification, they recommend evaluating the correspondence between fish assemblages and habitat classes defined on the basis of physiography and stream order. Finally, they recommend testing the classification with an independent dataset.

Evaluation: Classification according to the underlying processes or drivers of variability (e.g. climate and geology) facilitates a greater understanding of the systems being classified, and this is essential for effective management. Though the review by Portt et al. (1989) is dated, their analysis was very thorough and well thought through. They present an argument against the then traditional approach of determining the physical and chemical characteristics of the habitats in which biotic communities occur. They suggest that the traditional approach is problematic due to temporal variability – making habitat classification too unpredictable. Their suggestion that it is more practical to use the fishes themselves to predict habitat than the other way round was reasonable at the time for southern Ontario, but this would be difficult to do in areas of the Province that are data poor (e.g. the Far North). Portt et al. warn against the risk of misinformation due to multi-colinearity where causal connections can mistakenly be attributed between biota and abiotic variables even though the relationship is actually with a correlated variable.

Physiographic regions may reflect and capture glacial history, which is known to influence fish distributions (Jackson & Harvey 1989, Mandrak 1995, Neff and Jackson 2011). Adding network analysis to the geoclimatic approach may be a welcome addition to further refining such a system at a variety of scales.

Mandrak's (1999) Ontario wide aquatic classification was largely focused on establishing a correspondence between aquatic eco-regional groupings and fish faunal regions. Variables were selected based on hydrological processes thought to structure aquatic ecosystems at regional scales (i.e. bedrock geology, climate, glacial history and surficial geology). Climate and bedrock geology were expected to help explain fish distributions because both are related to the structure of fish distributions at regional scales, given post-glacial dispersal (Jackson & Harvey 1989, Mandrak 1995); and both climate and geology were used in the U.S. Nature Conservancy classification approach. Glacial history was derived from paleogeographic maps of glaciated North America (Dyke 1996) by determining the maximum area of glacial water coverage, total number of years deglaciated and glacial lake coverage within each watershed (14,000 to 6,000 years before present). Surficial geology was based on the Ontario Geological Survey (Barnett et al. 1991).

The framework proposed by Mandrak was based on multivariate analysis using clustering techniques, principal components analysis (PCA) and ANOVA. PCA was used to determine the dominant hydrologic process variables. Spatially constrained cluster analysis (spatially linked *k*-means clustering) was used to classify these variables into spatial groupings. The approach was to derive broad-scale classes and examine how they corresponded with fish distributions. ANOVA was also used to examine differences between resultant ecozones (5), ecoprovinces (10) and ecoregions (15) in terms of other variables like elevation, lake chemistry, morphometry, stream chemistry and stream flow. To compare the classification with known fish distribution patterns, Mandrak created a binary species presence/absence matrix based on data from the Centre for Biodiversity and Conservation Biology. Spatially constrained cluster analysis was used again to identify fish faunal regions based on similarity measures (i.e. coefficients of co-occurrence similarity and association similarity). Then results of the two cluster analyses were compared with each other by determining the areal overlap between the two maps (using a coefficient of areal correspondence).

Evaluation: Mandrak's aquatic eco-regions were based largely on similarities in geology and climate. Glacial history did not turn out to have as great an influence on the analysis as expected, and the authors speculate that this may be related to the fact that glacial history is highly correlated with many climatic variables. Also surficial geology was not as important as had been expected, but this may have been an issue of scale (their units were perhaps too coarse). Mandrak's emphasis on finding an areal correspondence between fish faunal regions and aquatic eco-regions is problematic because the results of his areal overlap analysis were used to determine the number of classes in the final map. This biases the results towards fish species distributions. According to Mandrak (1999), the five aquatic ecozones he identified are generally similar to the three terrestrial ecozones identified by Wiken (1986). But the 15 aquatic eco-regions he identified corresponded roughly with fish faunal regions and deviated substantially from terrestrial counterparts. Though Mandrak recommended that the next step in the AEC process for Ontario was to develop the classification at lower levels, work on the classification was not directly pursued within the OMNR.

The Great Lakes Conservation Blueprint project was completed in 2005 and as mentioned in the introduction, a key component of this project was an aquatic ecosystem classification for the region. The Great Lakes aquatic ecosystem classification (GL AEC) has five hierarchical levels that are almost all completely spatially nested, including: entire basins or primary watersheds (Fig. 3.5a); individual (Great Lakes) watersheds (Fig. 3.5b); ecological drainage units (Fig. 3.6a); tertiary watersheds (Fig. 3.6b); and aggregations of aquatic ecological units (Fig. 3.7). Each level in the hierarchy was designed to capture key processes operating at a scale thought to influence species distributions. For example, glaciation, speciation, and extinction processes were hypothetically portrayed by upper and mid-levels, genetic variation was hypothetically portrayed by mid-levels, and population dynamics were intended to be described by lower levels in the hierarchy (Wichert et al. 2004, after Frissell et al. 1986). According to Wichert et al. (2004), upper levels were defined using fish community patterns as key biological criteria (corresponding to primary watersheds throughout North America, as per Maxwell et al. 1995, Abell et al. 2000, and Mandrak 1999).

Aquatic ecological units (AEUs) are the smallest units in the GL Blueprint classification and they represent Great Lakes coastal polygons, inland lakes, wetlands, or stream segments. A water-flow layer created by WRIP (the Water Resources Information Project) was used as the stream backbone and ALIS (Aquatic Landscapes Inventory Software) was adopted from Michigan researchers (e.g. Seelbach et al. 1997) and used to break the water-flow network into segments. Segments were delineated based on changes in surficial geology, stream size, connection to other water bodies and the presence of barriers to flow. The drainage area around each stream segment was characterized during this process by summarizing geology, gradient and water storage potential in the surrounding reach or segment contributing area.

The Blueprint classification is a descriptive attribute driven classification, so no clustering or analysis was done. It was intended to be a practical tool structured around available data within the Great Lakes region. Aquatic units were placed into classes by aggregating them on the basis of similarity in selected characteristics (Wichert et al. 2004): stream segments were aggregated according to stream order or watershed position (headwater, middle, mainstem), gradient (gentle, medium, steep), geology (permeability to cold water species on a scale of 1 to 3) and water storage potential. Water storage potential was a function of the proportion of the drainage area upstream of a segment in the lake or wetland cover class (< 10% or >10%). Lakes, wetlands and coastal areas were summarized for different characteristics (e.g. size, shape, connectivity to streams or tributaries), in addition to permeability of the geology.

Over 120 aquatic classes can be recognized in the classification: there are 54 unique stream classes, 36 lake classes, 12 wetland and 27 coastline types (Fig. 3.7). These classes can be found anywhere in the Province; hence a headwater stream with high permeability, steep gradient and low water storage potential can be found in several areas of the Province (Fig. 3.7). The final Blueprint map shown in Figure 3.7 shows a combination of stream, wetland, lake, and coastline classes where wetland classes are shown on top of lake classes, lake classes supersede coastline classes, and stream or reach-contributing-area classes are depicted on the bottom (Staples 2004).

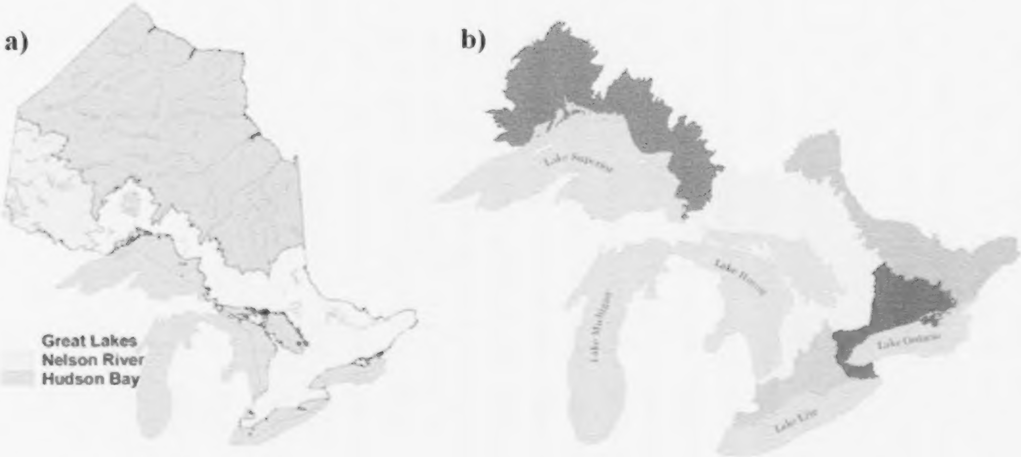
a)
b)
Great Lakes
Nelson River
Hudson Bay
Lake Superior
Lake Huron
Lake Michigan
Lake Ontario
Lake Erie

Figure 3.5. a) Primary watersheds of Ontario; b) individual Great Lakes watersheds

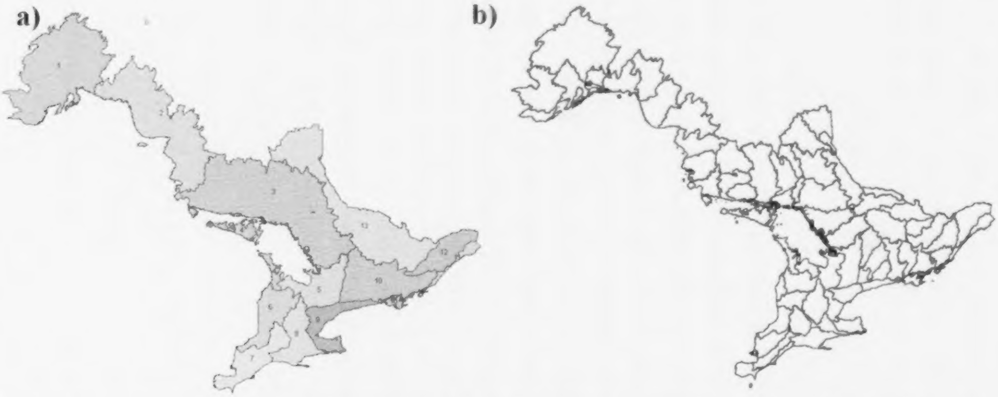

Figure 3.6. a) Ecological drainage units of the Great Lakes Blueprint aquatic classification; b) tertiary watersheds within the Great Lakes primary watershed.

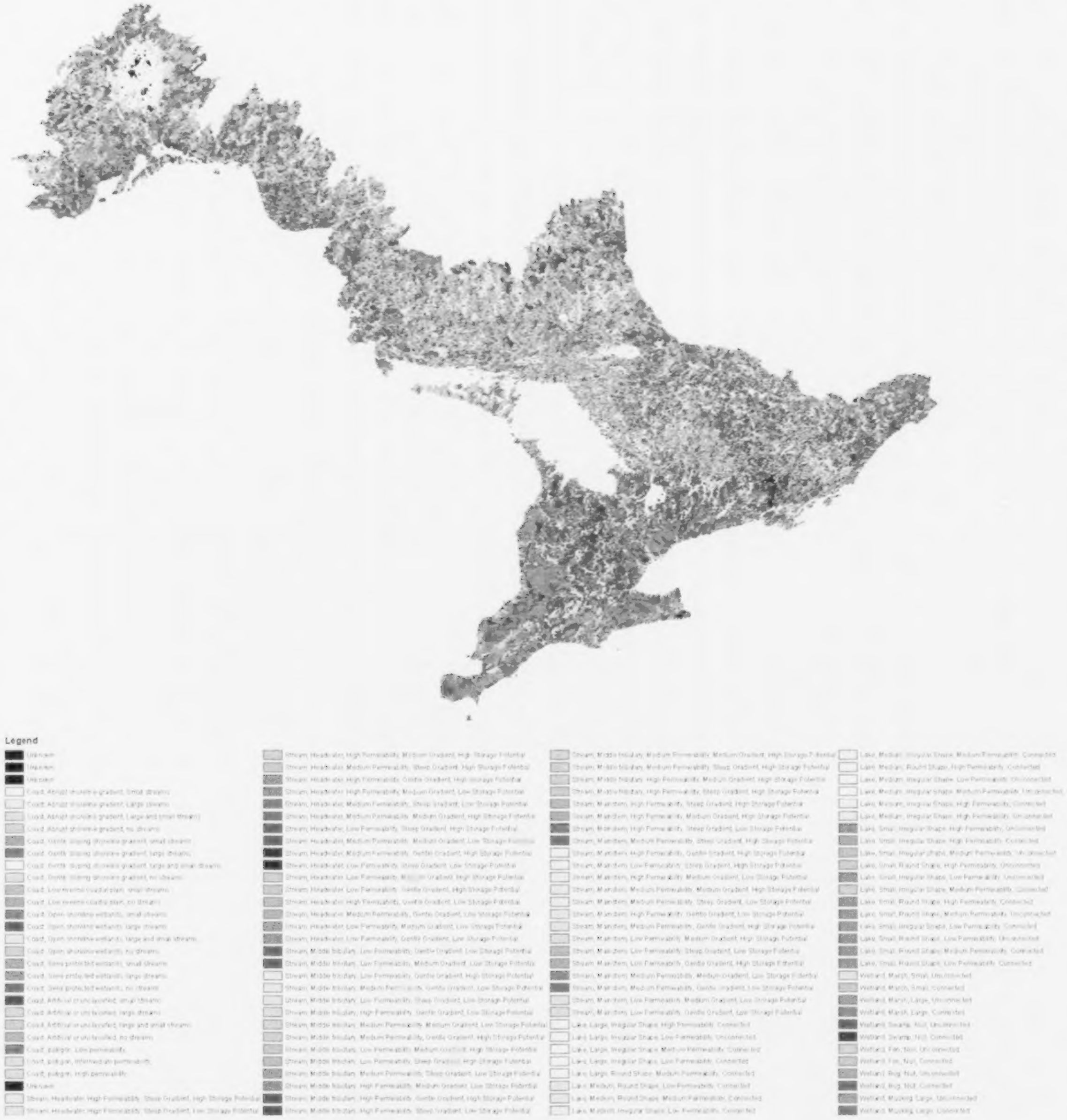

Figure 3.7. GL Blueprint aggregations of aquatic ecological units into ~120 different aquatic ecosystem classes, which represent stream, wetland, lake, or coastline types. Note that wetland classes supersede lake classes, lake classes supersede coastline classes, and stream or reach-contributing-area classes are on the bottom (Staples 2004).

Evaluation: The GL AEC was completed within a limited time period (~2001-2005) in cooperation with Nature Conservancy Canada, OMNR and the Natural Heritage Information Centre. The overarching goal of the Blueprint project was to produce aquatic and terrestrial conservation targets in the Great Lakes region of concern in order to protect the numerous and increasing number of aquatic species at risk in the region. To that end, the GL AEC is used to help characterize tertiary watersheds in the region by detailing where and what types of aquatic systems are located within them, how much of these areas are currently protected in parks and protected areas and which systems are under-represented. Detailed maps of tertiary watersheds were produced for the Blueprint project showing all aquatic ecosystem unit classes within each watershed and summarizing their conservation status and targets. These tertiary watershed maps are used effectively in combination with gap analysis, species diversity assessments, and rare species reports, to target land acquisitions by Nature Conservancy Canada, World Wildlife Fund Canada, and Ducks Unlimited (Cindy Chu, NCC, personal communication; Mike McMurtry, OMNR, personal communication).

Ontario's Biodiversity Strategy (OMNR 2005) refers to the Great Lakes Blueprint as a means to focus biodiversity conservation activities in southern Ontario (Action 24), and OMNR stewardship coordinators currently use this classification as a framework for identification of priority areas for stewardship projects. It is used and compared with results from other natural heritage system analyses. Moreover, local-scale conservation efforts (e.g. in the Bracebridge area) have been developed using the Blueprint project results as a base (Mike McMurtry, OMNR, personal communication).

Applicability for an Ontario-wide AEC: There are some issues with the Great Lakes Blueprint AEC that make it difficult to simply extend the approach throughout the rest of the Province of Ontario. First, given the current emphasis within the OMNR on biodiversity, the Blueprint's focus on aquatic regions that correspond primarily with fish faunal distributions may be insufficient when other taxonomic groups with various dispersal mechanisms are considered (e.g. aquatic plants and invertebrates). The protection of aquatic class parks and protected areas across the province aims to consider ecosystem level processes and multiple taxonomic groups.

Second, the Blueprint classification does not adequately capture climatic variation or longitudinal patterns in stream temperatures (Chu and Jones 2010). At the scale of aquatic ecological units (Fig. 3.7), the GL AEC ranked third, after Ontario Provincial and Canadian National land-based ELC's at eco-district scales, in terms of its ability to capture variation in stream temperature (Chu and Jones 2010).

Thirdly, many of the attributes selected to classify lakes and wetlands (e.g. connectivity to the water-flow network, yes/no; geologic permeability, high/medium/low; watershed storage potential, high/medium/low) were defined by expert opinion. Geologic permeability or 'permeability to cold water species' is a subjective measure of the relative contribution of precipitation to ground and surface water, which is based on percent cover of surficial geology (i.e. cover of well, moderate, and poorly drained soils) in upstream catchment areas. The significance of these variables, their influence on aquatic processes, and the methods used to derive classification cut-points were not adequately described or referenced by Wichert et al. (2004).

Wichert et al. (2004) note that EDU's (Fig. 3.6a) were based on drainage, geology and climate, and that the variance within EDU's was less than the variance between them, but methods and results for this analysis were not presented. Though EDU's integrated geologic and climatic factors, the way in which they did so was not analogous to terrestrial ecoregions because at least one EDU in southern Ontario crosses a major ecozone boundary (i.e. from the Mixed Wood Plain to the Boreal Shield) with largely different geoclimatic settings (e.g. EDU 5, Fig 3.6a). Moreover, although Wichert et al. (2004) state that EDUs were used to stratify AEUs at the segment level in order to account for larger scale processes such as climate and glacial historic processes, it is entirely unclear how this stratification was done. The analysis was restricted within the primary watershed boundary of the Great Lakes drainage basin (Fig. 3.5a), but no other stratification is readily apparent in the resultant classification (Fig. 3.7).

3.2.2 British Columbia

In the province of British Columbia, Miers (1994) and a task team developed a proposal for a hierarchical classification system based on physical (abiotic) aspects of the environment. The proposed classification system was based on the assumption that physical conditions in aquatic systems are related to the biotic response because different members of the aquatic community rely on different physical conditions in the stream for their survival, reproduction and dispersal habitat. Their goal was to create a classification system that would provide managers with guidelines for estimating habitat quantity, quality and suitability. The system was to be used for determining habitat protection requirements and for monitoring comparisons within and between classes over time. The first two levels of BC's terrestrial bioclimatic classification were also used in Miers et al. aquatic habitat classification (i.e. ecoregions and biogeoclimatic units) and were based on physiography and climate. Biogeoclimatic units were subdivided into terrestrial and riparian ecosystems. This was done for at least two reasons: 1) aquatic macrohabitat is 'ultimately determined by geology and climate' according to Minshall et al. 1985, and 2) this made it easier to standardize the terminology and intermesh the systems together. The finest level of resolution in the classification was designed to classify aquatic systems (wetlands, stream segments, lakes, estuaries and marine areas), but stream segments could be further subdivided into stream reaches, channel units and microhabitats. Moreover, lakes could also be further subdivided into open water and near-shore categories. Miers (1994) suggested that channel 'magnitude', summarized as the number of links draining into a particular stream segment, was a more reliable indicator of channel size than Strahler's (1952) stream order. Given the emphasis on abiotic factors alone, the system described in general by Miers is of interest for our purposes in Ontario, but the proposed system was never implemented nor adopted in B.C.

Later in the decade, Perrin and Blyth (1998) of Limnotech Research and Development (consultants) developed an ecozone classification for lakes and streams in British Columbia whose primary purpose was to provide a framework for setting water quality objectives and selecting reference sites for long-term monitoring of key water quality variables. Their classification grouped B.C. into three spatially-nested hierarchical classes: eight ecoprovinces based on geoclimatic factors; 45 ecoregions based on macroclimatic variation, characteristic lithology (rock and mineral compositional classes) and geomorphology; and 245 watershed groups, which were found initially by k-means clustering on the basis of various geoclimatic and chemical criteria such as TDS, pH, alkalinity, TP, true colour, TSS and turbidity. Watershed boundaries were established by the BC Provincial Ministry of the Environment Lands and Parks and they represent sub-basin catchments of a size appropriate for GIS analysis of large areas (i.e. boundaries were drawn for expedience). Due to a scarcity of data availability in the north of the province, watershed groups were eventually classified by expert opinion. Summary statistics of various water quality criteria were provided for each hierarchical level in the classification as an aid to water quality management planning. Whether or not this water quality classification system and associated ArcView database is or was used extensively is uncertain.

Most recently, Ciruna et al. (2007) of the Nature Conservancy Canada have developed EAU BC – a classification of ecological aquatic units in British Columbia. The system is based – to a large degree – on spatially-nested hierarchical approaches established by Higgins et al. (2005) of the Nature Conservancy. There are three nested levels in the hierarchy (ecoregions, ecological drainage units and river/lake ecosystems). The boundaries of the two upper hierarchical levels follow zoogeographic patterns in species distributions given post glacial expansion limits, current climatic constraints, physiography and established BC provincial third order watershed boundaries. These two levels are spatially-nested. But lower level groupings of the classification are place independent in that the 11 river ecosystem types, 23 river ecosystem sub-types and 9 lake types can be found anywhere within any ecoregion or drainage unit in the province. Similar approaches were employed by Snelder et al. (2004a) for New Zealand (see below). The smallest unit of analysis for lower levels are sub-watershed units from the watershed atlas of BC (>18,000 units), corresponding roughly to the river segment level in Seelbach et al. (1997).

Evaluation: The main weaknesses of this approach are:

1. The upper two levels in their hierarchy are limited to fish zoogeographic limits, which are not necessarily representative of the distributions of other taxonomic groups;
2. Little discussion of the conceptual basis for this classification is provided. Each level is really a separate classification. It is unclear how upper levels are used.
3. Mapping units were based on sub-watersheds from the atlas of BC, but there is no discussion of how watershed boundaries were created.

3.3 U.S. approaches

One of the earliest classifications in the U.S. was of Michigan streams (Anonymous 1967). This system was designed to assess the human-use value of surveyed streams. Field-based surveys were performed to determine three basic attributes that summarized 1) stream type and 'quality' (i.e. was it fishable?), 2) stream size, and 3) the extent of building development in the surrounding area per mile of stream. These inventory classes were then used to help establish planning objectives such as: water quality standards, fishing regulations, recreational value and land acquisition planning. The Michigan classification was based on a previous classification of trout streams, which was revised to include warm-water streams and species. Though current river segment classifications reflect a shift in values towards viewing rivers as ecosystems, the focus on fish assemblages and physical features (i.e. the geomorphology of the stream and valley channel) has remained pervasive in many U.S. approaches.

In 1979, several agencies in the United States, including the United States Geological Survey (USGS), the Soil Conservation Service (SCS), Forest Service (FS), Fish and Wildlife Service (FWS) and the Bureau of Land Management (BLM) were involved in a Commission set up by the Office of Management and Budget to develop a land inventory that included a classification system. Lotspeich and Platts (1982) developed a simple conceptual classification system based on causal factors designed to avoid the use of specialized nomenclature, and designed to integrate the riverine system. They argue that classification systems should be based on 'natural' attributes of the land (process-based), rather than on human uses of the land. Though this avoids problems associated with unanticipated future uses of resources, this perspective fails to acknowledge that humans and other biota can be profound agents of change to both land and topography. Biota are part of the natural landscape, however the majority of classifications do not adequately address biotic components of terrestrial and aquatic ecosystems at an appropriate scale. The classification system designed by Lotspeich and Platts was strictly abiotic (geoclimatic) with five spatially-nested hierarchical levels, including: *domains* of large climatic units; *provinces* of large physiographic units; *sections* with uniform macroclimate, geology and lithology; *regions* of unique drainage patterns; and *landtype associations* with uniform mesoclimatic influences. Though their system was never implemented - possibly because of the pre-eminence of Bailey's terrestrial ELC - aspects of their conceptual model are of interest in the development of an Ontario-wide AEC, which must be strictly abiotic (at least at the uppermost levels).

In the 1990's, the USDA Forest Service initiated work on a national hierarchical aquatic ecosystem classification to be linked with the terrestrial ecological land classification framework. Maxwell et al. (1995) describe the objectives and approach of this classification of riverine, lacustrine and ground-water ecosystems. Their classification was based on biophysical factors displaying both low temporal variability at a given map scale and a relationship with the aquatic ecosystem patterns and processes of interest. Larger hydrologic units and ground-water regions (subzones and regions) were regionalized based on zoogeography with a focus on fish communities and their geoclimatic influences (e.g. post glacial dispersal and evolution) as well as physiography. Smaller aquatic systems nested within these larger units were classified based on biophysical criteria. Riverine systems were comprised of stream networks within watersheds that could then be divided into valley segments, stream reaches and channel units. Lacustrine systems included lakes, ponds and reservoirs classified largely on the basis of geology, hydrology, morphometry, physical, chemical and biological features. Ground-water systems were grouped into aquifers whose hydrogeologic factors affected ground-water movement. In total, Maxwell et al. (1995) describe a hierarchical classification with seven levels (subzones, regions, subregions, river basins, subbasins, watersheds and subwatersheds). But, riverine systems could be further subdivided into smaller units, such as stream reaches, and lakes could also be further subdivided. Maxwell et al. (1995) provide a useful and detailed review of variables thought to be most important to aquatic processes and patterns at all scales and units of analysis. Over 200 references were cited in their report. As with other spatially nested, hierarchical classifications (e.g. Higgins et al. 2005, Snelder et al. 2004b, Ciruna et al. 2007) there are two main drawbacks to this approach, were it to be applied in Ontario.

1. Conceptual limitations (i.e. potential misapplication of hierarchy theory)
2. Limitations associated with using fish zoogeography to restrict classification of lower levels

Seelbach and Wiley (1997) provide an overview of the MRI project in Michigan as of 1997. This project represents a significant departure from previous approaches in the U.S. that focused on developing a spatially-nested aquatic ecosystem classification (as in Maxwell et al. 1995) because the focus was now on developing an inventory of rivers with segment level information. The goals of the MRI project were to 1) assemble data on important landscapes, river channels and fishes; 2) describe broad-scale spatial variation in the river ecosystem and fish community patterns observed; 3) develop models to elucidate mechanisms behind the patterns observed for the purpose of understanding or prediction; 4) develop a system of ecological classification units for Michigan rivers; and 5) develop decision-making tools. Thus the project began with an inventory to determine what data were available and to collect new data on river characteristics and their associated fishes. These data were used to develop descriptive and predictive, correlation-based models of physical stream characteristics associated with fish communities. The ecological classification was developed to allow generalization of complex river ecosystems. As such, the Michigan river classification system classifies rivers hierarchically at three spatial scales; catchment, reach (~5-10 km); and immediate sampling site (100-300 m). They focused on intermediate scales in time and space and they argued that although fish populations vary seasonally and annually, overall community composition within river segments (2-40 miles in length) should be fairly stable over longer time frames (e.g. multiple generations).

Their approach to river classification is taken from the standpoint that river valley segments are a natural ecological unit of organization that reflects the spatial scale of many physical and biological processes occurring in river systems (Seelbach et al. 1997). Homogeneous valley segments were identified by interpreting available geospatial data for; breaks in slope, major stream network junctions, landform boundaries, riparian wetlands and changes in stream sinuousity combined with fish distribution patterns. A variety of features create boundaries between river valley segments. Though the presence of mid-river lakes and impoundments are not one of these features *per se*, Seelbach et al. (1997) note that mid-river lakes can have strong effects on downstream chemical, thermal and material regimes citing Ward and Stanford (1983), Statzner and Higler (1985) and Frissell et al. (1986). They examined maps of stream-lake networks for natural breaks and they sent maps to a group of experts to distinguish between features such as: ‘headwaters’ and the river, major tributaries, large lakes and wetlands and changes in channel sinuosity, but it appears as though this methodology was not formalized. They did, however, track network position using a sequence of numbers to indicate proximity of a segment to downstream source populations.

Ecological classifications of river systems have seldom been implemented across large regions and not at a scale that is relevant for management (Seelbach et al. 1997). Moreover, previous river and stream classification systems in Michigan were essentially based solely on the distribution and abundance of game fishes. Much of the primary literature on stream classification has focused on forested, mountainous areas (e.g. Frissell et al. 1986, Miers 1994). The approach by Seelbach et al. (1997) is an attempt to provide a more inclusive, holistic – or ecosystem-based approach to river classification, which considered the complete system.

One of the theoretical drawbacks of river valley segment classification is that segments are assumed to be directly linked to surrounding land cover conditions, but segment break points are determined by expert opinion in many cases. The Michigan Rivers system is also limited, like all other classifications, in that boundaries are placed where the landscape is actually a continuum. Another limitation to the system described by Seelbach et al. (1997) is that climate and bedrock did not feature prominently in version 1 of their classification though the authors recognized that these variables are fundamental to describing broad scale geographic patterns. Moreover, lakes were largely ignored, and the presence of a lake did not necessarily create a break in the available hydrologic network layer.

In 2002, Seelbach et al. were successful in obtaining an EPA STAR grant worth 842K to further develop an ecological classification system for rivers that could be used to assess biological impairment in watersheds of Illinois, Michigan and Wisconsin. The project is now complete and various publications are available (Link). The potential to use landscape-scale variables to estimate riverine conditions at finer scales was explored (Seelbach et al. 2001, Baker et al. 2001. 2003). The group developed an ambitious set of landscape-based statistical models to describe aspects of river flow regime, water chemistry, summer temperature regimes, channel form, fish, invertebrate and algal assemblages. They evaluated two main approaches to incorporate information from the surrounding landscape: local buffer data was used in multiple linear regression (MLR) to predict site characteristics within specific catchments or local and catchment data were grouped into classes or types and then regression modelling was used to predict expected characteristics of each type.

Seelbach et al. (2006) make a distinction between Frissell et al's (1986) concept of hierarchically nested stream segments whereby all stream segments are contained within the same sub-watershed and a system that differentiates between neighbouring segments based on differences in the areas *that they drain*. This view of nested and overlapping catchments is standard in hydrologic analyses, but is not standard in approaches based on hierarchy theory (Melles et al. 2012). Seelbach et al. (2006) go on to reason that the position of a river segment within a catchment provides a basis for inference that relates catchment characteristics to response variables measured within river valley segments. They further reason that adding information on fish species associations and the hydrologic regime to the more standard variables used in river classification (i.e. the geomorphic variables) improves the delineation and characterization of rivers. In their view, the first step to river classification is to develop a data rich information system that can be used for multiple purposes because classifications are generally customized by the question of interest anyway (Davis and Henderson 1978 *cited in* Seelbach et al. 2006).

As described above, the river valley segment classification of Michigan streams designed by Seelbach et al. (1997 and 2006) had several recognized weaknesses (Brenden et al. 2008a, b).

For example, the stream segment attribute classes were subjective; interactions among attributes were not considered; and delineation of valley segment boundaries was based on expert opinion so these were difficult to reproduce in other areas or states. In order to overcome some of these weaknesses, Brenden et al. (2008a) design a classification to more objectively summarize segment types based on observed physicochemical and fish assemblage attributes. River valley segments were again the basic unit of classification, but here these were delineated on the basis of spatially constrained clustering analysis using landscape and river channel attributes associated with stream reaches in the US National Hydrography Dataset (Brenden et al. 2006, Brenden et al. 2008b). Although this initial level of aggregation from stream reach to valley segment is not a recognized, spatially nested, hierarchical level in the Michigan classification, some of the same variables used to determine valley segments were also used in the more traditional (place independent) approach to classification (e.g. mean July temperature, catchment area and slope were used to group river valley segments into segment classes). In their discussion, Brenden et al. 2008a discuss some of the strengths and weaknesses associated with their approach. The beauty of their approach is its simplicity. Only three variables (predicted stream temperature, catchment area and reach gradient) were used to classify streams into two levels (including 10 'thermal or hydrologic segment types' and 30 'geomorphologic segment types'). The use of predicted stream temperature may be useful for an Ontario classification, but details of this methodology were not discussed by Brenden et al. (2008a, but see Wehrly et al. 1997).

3.3.1 The Nature Conservancy Approach

The Nature Conservancy's (TNC) approach to freshwater aquatic classification around the world is place-dependent (spatially-nested), hierarchical and uses a combination of abiotic and biotic variables (Higgins et al. 2005). Their system is designed to support terrestrial ecoregional assessments and generates coarse-filter conservation targets, which according to TNC, can then be used to help drive conservation priorities, direct subsequent inventories that assess biotic diversity, and further refine their classification approach. The TNC approach is to represent the biodiversity of an ecoregion using a coarse filter that captures ecosystems and a fine filter to capture a subset of species and communities. What makes their system unique is that it is based on readily available, mappable data, collected at a scale that is appropriate for ecoregional assessment. The TNC classification system has four spatial levels: 1) aquatic zoogeographic units are large scale (10,000 – 100,000 km^2) regions that do not always conform with water basin boundaries and are distinguished by continental and regional patterns in zoogeography – predominantly fishes; 2) ecological drainage units are defined using drainage boundaries (but do not necessarily represent true watersheds according to the authors). EDU's represent regional aquatic biodiversity at a scale of 1,000 to 10,000 km^2; 3) aquatic ecological systems are stream and lake networks, and 4) macrohabitats are river valley segments typically 1-10 km in length.

Macrohabitats are the basic unit for their 'bottom-up' classification approach, which uses unsupervised (automated) classification to map segments into stream and lake networks (level 3) on the basis of high quality hydrographic data and other relevant spatial data layers. For example, agglomerative cluster analysis based on Sorenson's distance measures and Ward's group linkage methods can be used to create draft level 3 groupings. These groupings are then expert reviewed. A top-down classification approach is also employed by TNC in cases where GIS data are insufficient. The top-down approach uses available information and expert opinion on hydrologic regimes, physiography, geomorphology, vegetation and drainage patterns to create the stream and lake networks of level 3. Please refer to section 3.2.2 (*Evaluation*) for weaknesses of this approach.

Quite recently, Soranno et al. (2010) adapt TNC's approach to suit individual management needs for aquatic classification. Their approach is unique in that they use Higgins et al.'s (2005) aquatic ecoregion categories as predictor variables. They apply their approach to the classification of lakes into groups that represent different levels of eutrophication (phosphorus and chlorophyll levels). Other predictor variables are selected to cover processes that operate at different spatial scales from regional to local (i.e. precipitation, baseflow, lake depth, catchment area, elevation and measures of human population density and land use). Their approach to aquatic ecosystem classification is loosely based on the principles of landscape limnology (i.e. the patch-mosaic model) and hierarchy theory. Classification and regression trees and Bayesian tree models were used to perform the classifications and the resulting groupings were generally spatially contiguous because ecoregion class turned out to be a significant predictor variable. A major weakness of their approach is that they present a methodology to build predictive classificatory models that provide little (if any) ecological insight into why blanket predictor variables like Higgins et al's ecoregion classes are more important correlates of lake phosphorus and chlorophyll levels than factors like runoff, sediment characteristics and human population density.

3.4 New Zealand & Australia

3.4.1 New Zealand

Biggs and colleagues (1990) published a series of nine papers in the New Zealand Journal of Marine and Freshwater Research detailing the results of a large program in New Zealand to classify rivers according to their hydrologic, biologic and water quality properties. They took a correlation based approach to develop quantitative links between the physical environment (i.e. the geology, climate, hydrology and water quality of river catchments approximately 120 km^2 in size) and the biota (including periphyton, benthic invertebrates and trout). These relationships were intended to help make predictions about potential biological responses to human impacts and environmental changes. Over 100 sampling sites across New Zealand were examined, reflecting a range of conditions in this mountainous country. In comparison to comparable catchments in the United States, New Zealand Rivers have very high flow variability and specific yield. The synthesis paper by Biggs et al. (1990) provides a summary of the results found in each of the individual studies to characterize rivers, including the following: flow variability and flow related variables, water quality, water clarity and colour, periphyton, invertebrates and trout. They note that all classifications of sites according to their relative attributes displayed a clear gradient in their structure. In addition, a comparison of classifications was made possible by creating clustering dendograms of biotic communities, but no clear similarities among the biological classifications were found, which they suggest indicates that a classification scheme cannot be based on biotic criteria alone.

In order to select the best subset of predictor variables from the available physical attributes, the authors determined those variables that differed most between clusters and then they used discriminant analysis to identify the 'best' predictors of sites. DF analyses were moderately successful in allocating sites to the correct groups at least 60% of the time, and predictive power was reasonable at a suite of independent sites (50-67% accuracy), but model sensitivity was limited. Only two variables were consistently good predictors across the studies: proportion of developed pastures and mean catchment elevation. Given that the degree of collinearity was not examined, some relationships may have been overlooked and as with all correlative models, the causal relationships between variables remain indeterminate. Overall, the New Zealand classification approach points to a number of key variables that would be useful in an Ontario classification, namely: elevation, conductivity, flow variability (coefficient of variability), water temperature, landcover and catchment geology (refer to their Table 4 and Figure 1 for details). However, models that result from the correlative modelling approach are not likely to prove useful in the long-term without a more mechanistic understanding of the factors that control biotic communities. The New Zealand example is also likely to be a 'best case' scenario for a classification system because of the strong gradients that exist in this mountainous region. Classifications are likely to prove more difficult in areas with less topographic variability. The analyses techniques used by Biggs et al. (i.e. DF and clustering techniques) strongly reflect analyses that were popular during the late 1980's and early 1990's.

Two reports (Snelder et al. 2003 and Snelder et al. 2004b) and a fairly highly cited paper (Snelder & Biggs 2002) describe New Zealand's recent River Environment Classification (NZ REC) system in detail. Snelder & Biggs (2002) and the overview report provide the conceptual background and summary of the NZ REC, whereas the report is written as a user guide describing the REC in more detail and specifying: the conceptual underpinnings; how to use the NZ REC within a GIS; and how the NZ REC was made in technical detail.

New Zealand's REC is a hierarchical classification system with six levels covering successively finer spatial scales, from macro (105 km^2), meso (102 km^2), to micro (1 km^2). The basic unit of classification is the river section or segment (average length = 740 m), and river sections are described by the characteristics of their upstream catchment areas. Classification is performed by assessing a suite of six 'factors' (climate, topography, geology, land cover, network-position and valley landform) summarized within the upstream catchment area of each river section, or within the river section itself. Each level in the bottom-up hierarchy is effectively classified independently of other levels using environmental variables that describe a single 'controlling factor' known to be related to ecological variation at that scale. A large number of potential classes can be used to identify differences between river sections. For example, the controlling factor of geology has six categories, and the controlling factor of land cover has eight. This leads to 48 unique types of rivers from these two factors alone. It is readily apparent that with six factors and several categories within each, the number of potential stream types is unworkable for many management applications. However, Snelder and Biggs (2002) reason that the user needs to simply select the scale and resolution (i.e. the hierarchical level) that best suits the analysis. Moreover, they reason that 'source of flow', which is a measure of topography taking stream origin into consideration (e.g. glacial mountain, lake, wetland, low elevation, etc.), is a good starting point for defining management units.

Though the New Zealand approach builds on the principles of regionalized classifications, their approach differs in that ecological variation at each scale is addressed independently. They do not assume that total regional variation is captured by upper levels in the hierarchy as others do (e.g. Hawkins et al. 2000, Higgins et al. 2005), but they suggest that some ecological patterns and processes show significant variation within an ecoregion, depending on the 'controlling' process of interest (Snelder & Biggs 2002). This effectively makes the classification system a non-regionalized, place-independent classification. Classes with similar ecosystem types may show wide geographic dispersion. However, they write that the theoretical basis for the NZ REC assumes controlling factors act in a hierarchical fashion: lower levels in the classification (e.g. land cover) are determined in part by controlling factors at upper levels (e.g. climate). Another feature of the New Zealand approach is their application of the variable, 'network position' to aid in classifying river sections at finer spatial scales. Network position is summarized by assessing the Horton-Strahler stream order and categorizing network position into low order streams (order 1 and 2), mid-order streams (order 3 and 4) and high order streams (>4). Their approach was used by Wichert et al. (2004) for the Great Lakes Blueprint aquatic classification.

In 2007, Snelder et al. describe a method for selecting input variables, transformations and weightings that optimize the performance of an ecosystem classification. They argue that classification performance is sensitive to choice of input variables, and that their method is an improvement over the subjective selection of input variables. An iterative selection procedure was used to select environmental variables, weights and transformations. Next, Mantel tests were performed to maximize the correlation between the environmental and biological distance matrices for a subset of sites: the suite of variables with the highest Mantel correlations was selected. Clustering of the full set of river sections in New Zealand was then performed in two stages to accommodate data analysis size restrictions. First a non-hierarchical clustering procedure was used to define 500 groups based on their environmental distances from each other, as measured by Gower's statistic. Second, a conventional agglomerative clustering procedure (flexible unweighted pairgroup method with averages; UPGMA) was used to define relationships among the 500 clusters created by the initial non-hierarchical clustering. Classifications created in this way were better able to discriminate variation in river fish communities than subjectively chosen variables. Their findings suggest that classifications can be improved when variables with nonlinear relationships with the biota are transformed to improve the linearity of the relationships between biological turnover and environmental factors. Moreover, environmental variables with particularly strong relationships with biotic variation can be emphasized by weighting these variables more highly to improve resultant classification success. Overall, the 2007 paper is really a methods paper though they do present a classification showing two potential 'levels' with 5 or 10 classes respectively. Each classification level shows groupings of river segments based on a set of variables selected from a candidate set including: catchment slope, winter and summer temperature, the lake index, phosphorus, precipitation, elevation, distance to the coast and average and maximum segment slope.

This group of researchers in New Zealand led by Snelder are highly prolific at producing papers on river classification systems. In addition to the articles cited above, Snelder has been the lead author on at least six other papers on the subject of aquatic ecosystem classification. Many of these papers deal with methodological aspects of creating environmental classifications. But some of these methods contrast fairly sharply with the approaches actually implemented by Snelder and Biggs (2002) in their river environment classification system. The actual REC in New Zealand takes a far more subjective and qualitative approach to classification.

3.4.2 Australia

A geomorphic approach to river classification was taken in New South Wales, Australia that included aspects of river morphology (slope, valley shape and degree of valley confinement), the flow regime, sediment character and riparian vegetation. Geomorphic processes create the physical template of a river system, which then forms the basis upon which biotic communities rely and interact (Brierley & Fryirs 2000). Their approach was based on a modified version of Rosgen's river classification (Rosgen 1994, 1996), but they extend Rosgen's approach to include spatial (e.g. catchment) and temporal characteristics (Brierley & Fryirs 2000; River Styles). They describe a spatially-nested hierarchy of scales (after Frissell et al. 1986) as follows: catchments, landscape units, river styles, and geomorphic units. Examples of geomorphic units are things like differing types of bar, sand sheets, pools, riffles, benches, levees, backswamps, valley fill, terraces, etc. The proposed geomorphic approach uses the physical appearance of a river at differing positions throughout the catchment to provide an indication of its behaviour and habitat availability. They state that river behaviour is a reflection of landscape controls, dominance of bedrock, relief, material, and flow variability, as well as human impacts.

Thomson et al. (2004) tested the ecological significance of Brierley & Fryirs (2000) River Styles Framework by comparing macro-invertebrate assemblages and habitat characteristics of pools and runs in three different river styles in New South Wales, Australia. They found significant differences in the macro-invertebrate communities, particularly in pools, and they relate these differences to reach-scale substrate and flow characteristics. Differences in macroinvertebrate assemblages within the same river style were related to local habitat variables rather than to larger scale factors, consistent with other studies of macroinvertebrate community structure. Local factors are often more important than regional factors. They reason that knowing the large-scale context may not improve the ability to predict (macroinvertebrate) community structure if local habitat characteristics are not influenced by larger-scale variables. But, river styles based on geomorphic characteristics or local habitat variables alone are likely inadequate; they suggest including large-scale variables reflecting stream size, temperature and the hydrologic regime.

Other research groups in Australia have taken different approaches. For example, the Australian river assessment system (AUSRIVAS) is based on principles and methodologies developed in Europe for the River Invertebrate Prediction and Classification System (RIVPACS, Marchant et al. 1985, 1997; described below in Section 3.6 and in Melles et al. 2012). RIVPACS uses observed richness of macroinvertebrate communities in relation to expected richness derived from measurements at unpolluted reference sites (Simpson & Norris 2000, Turak et al. 2011).

Thompson et al. (2006) focused on classifying mountain streams in southeastern Australia using a geomorphic approach following and extending the work of Montgomery and Buffington (1997, described in section 3.1 above). They examined the role of geologic history and lithology for their influence on sediment supply, caliber and shape as well as resultant transport conditions. The authors describe eight dominant reach morphologies. Notably the authors found no significant longitudinal arrangement in the observed morphologies likely because increases in catchment area in their study systems did not result in commensurate decreases in slope, contrary to the norm.

3.5 Europe

Europe has followed a somewhat different trajectory in their literature on stream assessments, and spatially-nested hierarchical classifications are not necessarily the norm there (Melles et al. 2012). European countries have a long history of field-scale assessments of biological assemblages in streams and these assessments have been used for over a century to provide an indication of stream condition (Furse et al. 2006ab). European classifications tend to couple biological and physical features (Naiman et al. 1992, Wright et al 1998) and are based on the concept of a reference condition (circa Hughes 1995 *cited in* Furse et al. 2006b). Deviation from the reference state is used to determine site quality. These classification schemes (e.g. RIVPACS) assume that the relationships between stream biota and physical processes (geomorphic and hydrologic drivers) are predictable (Naiman et al. 1992, Wright et al 1998). Benthic macroinvertebrate assessments are by far the most common approach to evaluating European stream sites (Furse et al. 2006b). But, field-based assessments of biotic communities are time consuming and require a considerable amount of expertise, for example to identify invertebrates. For broader applications these systems should incorporate the physical features of the surrounding landscapes or catchments in order to make inferences about effects of surrounding land-use change related to water quality and in-stream physical habitat (Naiman et al. 1992, Wright et al. 1998, Davy-Bowker et al. 2006, Council of European Communities 2000).

Overall river classification systems developed in the UK are fine-scale, field-level assessments of aquatic communities and river environments. Raven et al. (1998) reviewed a variety of river environment classification systems in the United Kingdom in a move to develop a more integrated approach to support principles of sustainable development and biodiversity. These authors concluded by recommending the development of a hierarchical nested approach, integrating geomorphic processes at different scales to account for the range of organisms in aquatic systems. The push in the UK is to have a systems-oriented model that could characterize hydrologic, hydraulic, physical and chemical parameters associated with biological communities.

3.6 South Africa

Wadeson (1999) proposed an abiotic hierarchical model for South African river systems based on geomorphological characteristics. Their classification of streams was based on a modified version of Frissell's (1986) model. Their suggested model had six nested levels: drainage basin, stream segment, zone, channel reach, morphological unit (e.g. riffles and pools) and hydraulic biotope (e.g. habitat characteristics). Characteristics of the drainage basin such the biogeoclimatic region in which streams reside, the slope and shape of the longitudinal profile and indices of network structure would be used to classify entire basins. The hydraulic biotope was described as the scale that provided an essential link between geomorphology and ecology. Both morphological units and habitat characteristics formed the basic building blocks of their classification. The classification was both bottom-up (based on morphological units) and top-down because the channel network and catchment areas were viewed as complete units.

The approach to stream classification proposed by Wadeson (1999) was outlined as a future delineation exercise in an ecoregional river classification system for South Africa. Essentially their system is based on the principles of terrestrial ecoregion classification set out by Omernick in 1987 (Kleynhans et al. 2005). Relatively homogeneous regions, similar in terms of their physical, abiotic (i.e. climate, soils, geology, vegetation and physiography) attributes, are essentially sketched by expert judgement to delineate boundaries between ecoregions. The same physical attributes are used to delineate various levels in the hierarchy, but the relative importance of these attributes is allowed to vary from one place to another. Though terrestrial characteristics are used, the objective was to arrive at river ecoregions at two levels of detail, broader ecoregions and subcatchment (park reserve size) regions.

3.7 Lake and wetland classifications

Cowardin et al. (1979) developed an aquatic classification system that specifically included lakes and wetlands. These authors took a place independent, approach whereby units (i.e. the lakes, rivers, wetlands, marine ecosystems and estuaries) were not assigned to particular regions. Rather, ecosystem units within the same class could be found anywhere within the United States. However, Cowardin et al. (1979) recommended Bailey's (1976) terrestrial land classification system as a way to orient any aquatic system in the U.S. within a geoclimatic regional framework. Within Cowardin et al.'s classification system, aquatic units were classified into five different types (wetlands, lakes, marine areas, estuaries and rivers) according to their size, salinity, vegetative and physical attributes (e.g. confined by a channel). These aquatic units were further categorized into subsystems, classes, subclasses and dominance types with modifiers. Dominant life forms (e.g. trees, shrubs, emergents) and dominant plant and animal species were emphasized in their classification (*circa* Clements 1916). Interestingly, the authors had to deal with the issue of semantic ambiguity because familiar terms such as marsh, swamp, bog and meadow were not consistently used and applied across various regions of the U.S. An advantage to Cowardin's classification is that it is a more flexible approach than regional classifications. Aquatic systems in different areas of the continent may have similar ecological drivers and developmental contexts even though they are not necessarily in the same spatial region.

A very useful schematic model for developing a classification system for lake habitats is presented by Sly and Busch (1992). This model depicts the major physical and chemical interrelationships that control metabolism and productivity in a lake. A similar approach has been suggested for rivers whereby river productivity can be modelled using physical and chemical variables (e.g. Welcomme et al. 1989), and the geographic location of lakes can be used as a modifying physical factor of the rivers themselves.

The system developed by Sly and Busch (1992) was designed to interface with the U.S. system for wetland and aquatic ecosystem classification (Cowardin et al. 1979). Each level in their non-spatial hierarchy was described by a basic set of data that were used to describe confining factors (morphology), forcing factors (light, heat, movement) and response factors (aquatic vegetation, sediment sorting). Aquatic units (lakes, rivers, estuaries) were grouped into systems based on flow characteristics, residence time and salinity. Systems were then subdivided into sub-systems (open water and near-shore), divisions (circulatory basin and entire shoreline), sub-divisions (particular features of shoreline or circulatory basin) and classes (water column, substrate and plant material). Miers (1994) suggested that this type of classification system was applicable to B.C. as well because it fit well with their developed list of aquatic plant habitats. The variables used as modifiers in Sly and Busch's system provide a useful description of factors that influence limnological processes.

Jaworski and Raphael (1979) classified coastal wetlands of the Great Lakes area on the basis of geomorphology. Coastal wetlands were classified as one of five types: A) sheltered lagoons bounded by sand barriers, B) bird foot deltas where sediments are deposited by a river, C) flooded river estuaries, D) shoreline complex wetlands, and E) coastal barrier wetlands that are bounded by spits, ridges or sand barriers. The key characteristics in this wetland classification were lake plain sediments, lagoon/wetland habitat/ natural levees, coastal barriers/dunes, lake terraces and glacial uplands. Also, Harris et al. (1996) developed a place-independent wetland ecosystem classification for northwestern Ontario

The OMNR (1984) developed a combined wetland classification and evaluation system by scoring wetlands (> 2 ha) based on their biological, social, hydrological and special attributes, for use in land use planning and habitat management. In this system, wetlands were assigned values based on wildlife habitat and their value to naturalist groups including educators, recreational users, scientists and local residents (McKee et al. 1992). Scores were then used to classify wetlands into seven ranked categories. Details of variables used to score wetlands within each component from biological to social are described by McKee et al. (1992). The biological component included variables that summarize productivity, diversity and size. The social component included a wetlands' cash value in terms of resources, its recreational, aesthetic and educational value, in addition to its proximity to urban areas, ownership and size. The hydrologic component included flow stabilization values, water quality values and erosion control value. Special features included an estimate of a wetlands relative rarity, significant features and ecological age – a function of its type (i.e. bog, fen, swamp, or marsh).

In 2000, the Great Lakes Coastal Wetland Consortium (GLCWC) (http://www.glc.org/wetlands/) was convened by the Great Lakes Commission to '*expand the monitoring and reporting capabilities of the U.S. and Canada under the Great Lakes Water Quality Agreement*', and they were given a budget of 1.2 million dollars to do so. The reader is referred to Albert et al. (2003, 2005) for details on recent developments in the Great Lakes Coastal Wetlands Classification. Additionally, the Great Lakes Blueprint Classification created 27 classes of coastal aquatic ecosystem units on the basis of shoreline gradient, the presence and type of wetlands (i.e. subject to wave action or not), and the size of streams draining at the coast.

3.8 Segmentation

All classification exercises begin by segmenting land or water into basic units or zones that are easily recognized, distinctive and homogenous in the characteristics of interest. With bottom-up approaches, an initial segmentation is done to identify basic spatial units (i.e. the uniform puzzle pieces) that can then be placed into groups. With top-down, or regionalized approaches the land area is subdivided into zones that are then further subdivided at finer and finer spatial scales and/or levels of detail.

Aquatic and terrestrial classifications generally begin with units that have recognizable boundaries around distinctive geomorphic formations such as lakes, wetlands and ridges or around distinct geologic, climatic and vegetative zones such as shield, tundra or boreal forest. In many cases there are obvious ecological implications associated with distinct boundaries: fishes cannot leave water; a sharp ridge is an impassable barrier to many land animals. In many other cases, boundaries are indefinite and the ecological implications of indistinct divides (or fuzzy ecotones) are not fully known. For example, although the importance of climate in limiting species' distributions has been confirmed by numerous studies, identification of climatic zones is fuzzy, problematic and correlative (Gaston 2003). We may have strong inference that temperature affects the distribution of some species: brook trout (*Salvelinus fontinalis*) for instance, shift their distributions to river reaches that are 200 m lower in elevation (on average) with every 1° increase in latitude (Flebbe 1994). But the ecological implications of indefinite climate boundaries are unknown for many organisms. Moreover climate is but one of many factors influencing species' distributions.

It is therefore easy to appreciate how identification of appropriate spatial units or spatial divides for classification is one of the most important and challenging aspects of ecosystem classification (Brenden et al. 2008b). Perhaps no universal system for distinguishing ecosystem boundaries can exist given the manifold nature of ecosystems of different size and the effect that a change in spatial scale can have on how we perceive them (Bailey 1987, Loveland and Merchant 2004). The challenge is particularly evident when it comes to identifying appropriate spatial units or divides for lotic ecosystem classification. Whereas the edge of a lake, pond or wetland may be easily identified, flowing water systems are made up of many interconnected stream reaches, creeks, small brooks, and hundreds of tiny rivulets that make it difficult to segment (Brenden et al. 2008b). Moreover biotic communities in rivers have long been expected to respond to longitudinal and directional changes in stream characteristics along a continuum with changes in flow, temperature, gradient, depth and organic matter (Vannote et al. 1980). Some of the foundational research on rivers led to classifications with overlapping zones along longitudinal gradients characterised by particular sets of environmental conditions and biotic communities (Hawkes 1975).

Current research in stream ecology suggests that tributary streams modify the river continuum and the magnitude of adjustment depends on tributary size relative to the main-stem as well as the physical, chemical and biological characteristics of the tributary (Bruns et al. 1984, Rice et al. 2001, Benda et al. 2004a, b). There may even be situations where it is desirable to consider important tributaries or confluence zones as separate and unique ecological units themselves because these areas are associated with increased productivity and they contain heterogeneous aquatic characteristics with unique morphology, hydrology, sediment fluxes and environmental conditions (Rice et al. 2008). River systems are now often understood as 'fluvial landscapes' with many relatively distinct and interacting patches (Lake 2000, Poole 2002); they are no longer viewed as simple linear continuums that can be divided up into zones; rather they are more often viewed as non-linear network systems with many small branches feeding into larger tributaries and eventually the main-stem (Benda et al. 2004a). In addition, river networks are often interrupted by lakes and impoundments in different positions, which further modify downstream conditions (Kratz et al. 1997, Riera et al. 2000, Martin & Sorrano 2006, Jones 2010).

Streams are generally segmented on the basis of bifurcations in the river network (Gravelius 1914, Horton 1945). Lakes and significant geological features such as narrows, rock transformations and waterfalls can also form breaks in these directed aquatic networks. What is more difficult to assess is whether these geomorphic stream breaks have ecological relevance. Stream segments can be delineated automatically using algorithms implemented in geographic information system software, and these methods have been used to segment streams based on confluences and changes in stream size, connections to other water bodies, the presence of barriers to flow, and intersections with features of surficial geology (Stanfield et al. 2006). However, automated approaches create many small and irrelevant stream segments, and there is little evidence to suggest that hydrologic flow (or other morphological and ecological processes) in a river change immediately or substantially at all confluences and at all changes in surficial geology (Benda et al. 2004b). It follows that identification of ecologically significant break points in aquatic networks is of crucial importance. How can we identify the confluences, tributaries, narrows and geological changes that significantly affect aquatic process regimes of interest (i.e. flow, temperature, sediment loading and nutrient cycling)?

Recently, Brenden et al. (2008a) proposed a new and more objective approach to delineate segments, which addresses this question and improves upon automated approaches. This is an active area of research. They developed a tool to link similar and spatially contiguous inter-confluence reaches together using a valley segment affinity search technique (VAST). VAST uses a clustering algorithm similar to *K*-means to create groups by adding inter-confluence reaches one at a time based on similarity criteria and spatial contiguity constraints. They used physico-chemical variables to create homogeneous linear segments in their network of Michigan streams, including $\log_e$ upstream catchment area, predicted temperature, percent lake, percent wetland, percent moraine, mean slope, and predicted baseflow. These authors suggest that river valley segments with homogeneous hydrology, limnology, water chemistry, channel morphology and riparian characteristics are a natural ecological unit of organization (Seelbach et al. 1997, Brenden et al. 2008ab). As such, river valley segments should reflect the spatial scale of many physical and biological processes occurring in river systems (Seelbach et al. 1997, Brenden et al. 2008a).

The point of any classification is to group similar units into classes that ideally provide an indication of how the systems under study behave. With ecosystem classification, we expect (or hypothesize) that each class responds similarly to stressors so that the classification provides some indication of how to manage and maintain different ecosystem types. An ecosystem can be defined as a unique and dynamic complex of plant, animal and micro-organism communities, as well as their non-living environment, all interacting as a functional unit (International Convention on Biological Diversity 1992). Most classifications rely on characterizations of the abiotic environment to provide an indication of related biotic communities that likely inhabit a given class (Fig. 3.3). Once ecologically significant aquatic segments are identified, a variety of clustering approaches can be used to create classifications of these ecological units.

However, there are a couple of issues that must be considered. A given segment of river, lake or wetland is not isolated from either its immediate local catchment area or its entire upstream drainage area. Larger scale processes like climatic cycles, geology, and topography help shape individual stream segments (top-down), but longitudinal up- or down-stream disturbance events can have profound influences on the properties of a segment as well, even if those disturbance events occurred at great distances away in either direction (Fausch et al. 2002). Moreover, local-scale processes such as riparian influxes can have an impact on a given section of stream too. Hence the question becomes, do we cluster segments of stream based on their local catchment characteristics (the isolated puzzle pieces)? Do we cluster segments based on their upstream catchment characteristics (the puzzle pieces linked with their contiguous upstream neighbours)? Or do we cluster based on large-scale factors like climate and geology in the broader surrounding area (grouped puzzle pieces of similar size)? Often some combination of all these factors characterized at variously nested spatial scales, regardless of their non-independence, is used to cluster river segments.

As you move down the network in the direction of flow, we know that the width and depth of river segments change. Mean annual discharge, channel size, alluvial habitat and contributing area all generally increase in a downstream direction, which results in predictive species-discharge relationships that are similar to land-based, species-area curves (Poff et al. 2001, Olden et al. 2010, McGarvey and Hughes 2008, McGarvey 2010). Headwater systems tend to be occupied by different suites of biota, such as colonising specialists and species with the ability to withstand more unstable environmental conditions, than downstream systems (McGarvey 2010, Finn et al. 2011). Finn et al. (2011) reasoned that such comparisons were made under the linear river continuum view of stream ecology. But a branching network view leads to comparisons of (beta) diversity across different headwater branches (Finn et al. 2011). Perhaps it is inappropriate to cluster variously sized segments together. Are headwater streams truly comparable with mid- and higher-ordered streams (Finn et al. 2011)? This seems analogous to clustering cycling paths with multi-lane highways. However, mid-order streams are connected to headwater streams and their upstream catchment areas contain headwater catchments so it also seems entirely reasonable to consider the entire network as a single system.

Eventually, managers and landscape planners are interested in what is going on in an entire watershed overall. Of course, the watershed will be composed of many different segment types, but can we make any generalizations about the class of an entire watershed? It is not necessarily sufficient to classify all segments separately without considering flow and connectivity within an entire watershed or basin. One of the issues we face is how to classify watersheds that are composed of very different stream types. Resultant watershed classes may contain more variation within the same watershed than between different watersheds. For example, headwater rivers, lakes, and wetlands in different basins often share more similar abiotic (and biotic) properties than do headwater and mainstem sections within the same watershed (Kratz et al. 1997, Fullerton et al. 2010, Finn et al. 2011). Even so, many land-water planners will be interested in the overall class of a watershed, which would summarize all upstream network and contributing area characteristics, and provide a picture of the entire watershed.

4 Limitations of aquatic classifications

Creation of ecosystem classes inherently implies that relatively distinct boundaries exist and can be identified by a discrete set of variables. The reality is that aquatic systems are innately complex with multiple branches interspersed by lakes and wetlands, longitudinal and lateral linkages to the surrounding landscape, and vertical linkages with groundwater aquifers that are generally difficult to observe. Aquatic systems change seasonally, yearly, and episodically with geo-climatic disturbances: they are flow connected. Hence the boundaries between apparent fluvial patches or zones are often indistinct (Naiman et.al.. 1988, Cleland et al. 1997). The intrinsic connectivity and variability of aquatic systems are fundamental and essential properties, but these properties make placement of enduring classification boundaries difficult and often subjective (Bailey 1983, Naiman et al. 1992).

An ecosystem classification or map represents an hypothesis about how we think ecosystems behave (Bailey 1983, Snelder and Biggs 2002). They assume that the boundaries we create represent meaningful differences to the targets of interest (e.g. species diversity, ecosystem productivity) because there are significant differences in geologic, climatic, hydrologic, nutrient and sediment regimes. Nevertheless, most classifications do not adequately model biotic processes that are also known to affect species distribution patterns (i.e. dispersal limitations due to current or historic barriers; competition from invasive species; and broad-scale post-glacial distributional changes). These biotic factors likely correlate with abiotic environmental features such as geology, but often there is not enough information to thoroughly model whether, for example, geology is a good surrogate for things like post glacial dispersal. Moreover, AEC's assume that ecosystems within the same class respond similarly to management actions (Bailey 1983). These assumptions need to be tested. Aquatic ecosystem classes are to be treated as hypotheses rather than paradigms.

Top-down regionalized classifications have a different set of limitations than bottom-up approaches. Regionalized classifications have been criticized for their inherent subjectivity. Though they attempt to capture top-down controlling factors and a sound theoretical understanding of aquatic ecosystem processes, mapped boundaries are often essentially sketched on the basis of expert opinion and knowledge of ecosystem characteristics (Hawkins et al. 2000, e.g. Omernik 1987, Kleynhans et al. 2005, Higgins et al. 2005). Hence they are rarely based on a mechanistic, process-based understanding of the drivers behind biological productivity and diversity (Hawkins et al. 2000, Young et al. 2002). Though the regionalized approach has intuitive appeal because it creates a simplified picture of aquatic productivity and diversity based on homogeneous and easily recognized units that can then be managed and reported upon, there is little evidence to suggest that aquatic systems are more fittingly grouped into spatially homogeneous regions than place-independent groups with non-contiguous boundaries (Jenerette et al. 2002).

A variety of clustering algorithms are available to perform bottom-up or place-independent classification once an aquatic network has been segmented into homogeneous units of the desired size and characteristics. Aspatial clustering methods include, *k*-means clustering and conventional agglomerative clustering procedures (e.g. the flexible unweighted pairgroup method with averages; UPGMA, Snelder et al. 2007). Clustering can also be performed under spatial contiguity constraints using Generalized Dissimilarity Modeling, Classification and Regression Trees, Random Forest models (Snelder et al. 2010), or *k*-means clustering adapted for spatial contiguity constraints (Mandrak 1999, Brenden et al. 2006, Brenden et al. 2008a). Many of these methods use a variety of multivariate variables that make it difficult to interpret the meaning of resultant classes. Indeed the complexity of some of these methods may hinder their explanatory power because it is difficult to establish causal linkages behind all of the correlations. Although classification accuracy rates can be assessed, this does not necessarily make it easier to understand why classification fails in some cases. Subjective decisions such as pre-selecting the number of classes to create are still often involved. Moreover, choice of a spatial framework for organizing environmental information is not trivial (as discussed in section 3.8). Selection of the basic unit: stream reach; surrounding or upstream catchment; river segment; entire lake and wetland or depth strata; sections of shoreline; or watersheds and basins of different sizes can have important implications for classification. The choice can influence the effectiveness of ensuing research, assessment and management of many aquatic-resource problems, leading to the generation of large amounts of information and economic expenditures (USGS 1995). Hence, it is imperative that the framework and units are comparable for them to be useful.

With approaches that use a mixture of both top-down and bottom-up methods (e.g. Perrin and Blyth 1999, Weitzel et al. 2003, Wichert et al. 2004, Seelbach et al. 2006), the result is that some levels have spatially continuous regions, whereas other levels have non-contiguous and overlapping groups. This makes it difficult to assess how different levels in the classification are linked, and whether or not each level has different theoretical foundations in aquatic ecology (e.g. are some levels longitudinally arranged as per Vannote et al. (1980), hierarchically arranged as per Frissell et al. (1986), or some undefined new approach that lies somewhere in between?).

Ultimately because all classifications result in a fixed map (or spatial model), all classifications suffer from their lack of ability to deal adequately with temporal change. However, inter-confluence reaches, segments, and watersheds should not change markedly over relatively short periods of time, and most classifications do not map ephemeral river characteristics that change with high frequency (e.g. riffles and pools). Ideally the variables used to describe ecosystem types will capture seasonal and longer term variability in key ecosystem process regimes. For example, the flow regime of a river over time may best be explained by a few key parameters such as flow predictability, flood frequency, flood predictability, and intermittency (Poff and Ward 1989). Models for key process regimes each have their own errors and uncertainties, and these may be quantified and carried through as ecosystem classes are modelled. Ideally, ecosystem class membership will be a probabilistic mapping exercise that must be calibrated and validated if it is to be used for prediction.

5 Conclusions and synthesis

Conservation and management of freshwater ecosystems involves evaluating and managing effects of cumulative impacts on the aquatic environment from complex pressures including; land use change, point and nonpoint source pollution, the creation of dams and reservoirs, forestry, mining, and fishing. To assess the impacts of these changes on associated biotic communities, it is necessary to monitor and report on the status of aquatic ecosystems. A variety of river and lake classification methods are available world-wide to assist with these tasks, and such methods attempt to provide a systematic approach to modelling and understanding complex aquatic systems at various spatial and temporal scales. Classifications create a typology of unique ecosystem types that can be mapped and further analyzed so that managers can target research and conservation efforts across large regions. However, a generally accepted approach to aquatic ecosystem classification is still forthcoming.

The approaches to AEC that we have reviewed can be grouped into three general categories (Melles et al. *in review*): 1) those that emphasize longitudinal patterns from headwaters to river outlets; 2) those that emphasize hierarchical controlling processes at different spatial scales using top-down and some bottom-up GIS analyses; and 3) those that emphasize site similarities between abiotic factors and/or biotic communities in a place-independent fashion, where geographic location and longitudinal position does not necessarily influence classification of a site. Few approaches account for both the directional nature of aquatic networks *and* regional environmental differences between catchments at different spatial scales in a process-based way. In addition, classifications typically fail to fully integrate flowing waters, lakes, wetlands, and groundwater aquifers. A more flexible approach to classification would enable a variety of classification 'views' to be explored depending on the purpose of the classification. Within watershed 'views' may emphasize longitudinal factors whereas between watershed 'views' may emphasize regional differences between aquatic systems of similar size.

6 Where to from here?

In a follow up report, we outline seven steps or considerations that may assist with the development of more flexible and integrated classification models based on current understanding, network thinking, and theoretical underpinnings (Melles et al. *draft technical report*). Based on our review of aquatic ecosystem classifications, we believe that an abiotic (physically-based) approach is most appropriate for Ontario. We recommend a combined geo-climatic (including climate, physiography and surficial geology) and hydro-chemical (including flow and water chemistry), network-aware classification. The classification would support an inventory database that can be used to generate multiple 'views' or maps given a particular user need. We will create a generalized classification view that should prove useful to many users. Such a classification should capture the key underlying process regimes that are critical to the functioning of aquatic systems (i.e. the flow regime, stream temperature regimes, and the nutrient and sediment regimes). A multi-scale classification with five levels is suggested, including aqua-zones, quaternary watershed collections, segment sets, reaches, and aquatic sites. Each level in the classification would be a stand alone classification designed to model key processes relevant to that particular spatial scale. The basic units of classification (e.g. quaternary watersheds, catchments) at different spatial scales should also be relevant for various land use planning, monitoring, and reporting purposes.

7 References

* Asterisked references are aquatic classifications that were summarized for general trends.

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